Migratory shorebird ecology in the Hunter estuary, south-eastern Australia

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DEDICATION

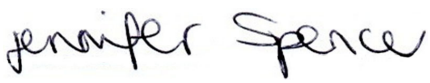
I dedicate this thesis to the memory of my father John

"A powerful agent is the right word: it lights the reader's way and makes it plain; a close approximation to it will answer, and much travelling is done in a well-enough fashion by its help, but we do not welcome it and applaud it and rejoice in it as we do when the right one blazes out on us."

MARK TWAIN

STATEMENT OF ORIGINALITY

I declare that this dissertation is my own work and has not been submitted in any form, fully or partially for a degree or diploma at any other university or institution of higher (tertiary) education. All information derived from the published and unpublished work of others has been acknowledged in the text and a list of references is provided.



Jennifer Spencer, February 2010

ABSTRACT

Migratory shorebirds inhabit the shorelines of rivers, wetlands, oceans and lakes, where they need to rest and feed during their non-breeding seasons to prepare for their annual migrations to breeding grounds in the Arctic. Along their flyways many non-breeding and stop-over sites are under pressure from coastal developments, disturbance, global sea level rise and water resource development. In this thesis I investigated how migratory shorebirds responded to habitat loss in the Hunter estuary, a non-breeding site in south-eastern Australia, and how they used remaining estuarine habitats.

The Hunter estuary is a wetland of international importance but has a long history of modification from industrial and urban development which began in the late 1800s. Based on recent counts (2001-07), the Hunter estuary now only supports two species in internationally significant numbers (Eastern Curlews *Numenius madagascariensis* and Sharp-tailed Sandpipers *Calidris acuminata*) compared to seven species listed from earlier records (1970-90). Overall, I detected a 42% decline in total numbers of migratory shorebirds (1981 - 2007) in the estuary, with significant declines (44 - 83%) in four species.

The tidal cycle was the main driver for the distribution of shorebirds in the Hunter estuary with most species feeding in exposed intertidal mudflats at low tide and being forced to rest at high tide. Eastern Curlews roosted on artificial structures and sandbanks during the day but at night they moved to flooded saltmarshes. Shallow water was important at the roost sites, as it provided a mechanism for cooling on warm days and for detecting predators at night. Vigilance behaviour made up 30 - 40% of Eastern Curlews roosting time, but vigilance increased by about 20% prior to their migration (Feb-Mar). Day roosting habitat was most limited during spring high tides and periods of high disturbance. Disturbance was significant at the day roosts (0.8 - 1.7 hr^{-1}), mostly from birds of prey, but Eastern Curlews spent longer in flight after being disturbed by people.

The tidal period and tide type (neap or spring) determined shorebird distribution on intertidal mudflats. Intertidal mudflats in Fullerton Cove provided important feeding habitat for many species, but artificial mudflats impounded in the North Arm of the Hunter River extended foraging time for small shorebird species which fed until the tide forced their relocation to the main day roosts. These impoundments increased in importance for all shorebirds during neap cycles and one to three hours before high tide when the availability of intertidal mudflats was limited in the rest of the Hunter estuary. Overall, a high percentage (> 90%) of Bar-tailed

Godwits *Limosa lapponica* foraged during low tides and 50% of godwits continued to forage up to three hours after low tide. Foraging Bar-tailed Godwits were most successful in mudflats in Fullerton Cove but prey availability was not uniform among mudflats.

Saltmarshes provided major night roosting habitat and important feeding habitat for small shorebird species, including Sharp-tailed Sandpipers, which foraged in saltmarsh regardless of the tidal period. Generally, most shorebird species avoided saltmarshes with large stands of mangroves, preferring sites with open saltmarsh and tidal pools.

Changes in attitudes towards wetlands management in the last three decades coincided with the formal protection and rehabilitation of shorebird habitat in the Hunter estuary. In 1995, culverts were removed to restore tidal flushing to estuarine wetlands on Ash Island, a highly modified wetland complex in the Hunter estuary. Although high inter-annual variability in migratory shorebird populations made it difficult to detect short-term responses to wetland manipulation, long-term monitoring (1994-2007) indicated that increased tidal flushing had promoted mangrove expansion indirectly reducing habitat availability for shorebirds. Mangrove removal has the potential to restore this imbalance, but further studies are needed to support an adaptive management approach to managing shorebird habitat in the Hunter estuary.

The cumulative loss and degradation of estuarine habitats in south-eastern Australia and other parts of the East Asian-Australasian flyway continue to threaten shorebird populations, but these impacts could be addressed through greater commitment to the protection and active management of shorebird roosting and feeding habitats in their non-breeding range.

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PUBLICATIONS RELATED TO THIS THESIS

- Spencer, J.A. and Kingsford, R. T. (*in review*). Observer error in shorebird counts. (Appendix A1).
- Spencer, J. A., Monamy, V., and Breitfuss, M. (2009). Saltmarsh as habitat for birds and other vertebrates. Chapter 7 In '*Australian Saltmarsh Ecology'*. (Ed. N. Saintilan). pp. 149-165. (CSIRO publishing: Collingwood, Victoria). (Appendix A2).
- Howe, A. J., Rodríguez, J. F., Spencer, J.A., MacFarlane, G.R. and Saintilan, N. (*in press*). Response of estuarine wetlands to reinstatement of tidal flows. *Marine and Freshwater Research*.

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LIST OF ABBREVIATIONS

ABBBS	Australian Bird and Bat Banding Scheme
ACU	Australian Catholic University
ANCOVA	Analysis of Covariance
ANOSIM	Analysis of Similarity
ANOVA	Analysis of Variance
AWA	Australian Wetland Alliance
AWSG	Australasian Wader Study Group
BACI	Before and After Control/Impact
BIOENV	Biological-Environmental linkage procedure
BOM	Australian Bureau of Meteorology
CAMBA	*China-Australia Migratory Bird Agreement*
CI	Confidence Intervals
CMA	Catchment Management Authority
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CVA	Conservation Volunteer Australia
DECC	Department of Environment and Climate Change (NSW)
DEH	Department of Environment and Heritage
EAAF	East Asian-Australasian Flyway
EAASSN	East Asian-Australasian Shorebird Site Network
EEC	Endangered Ecological Community (*TSC Act*)
EPBC Act	*Commonwealth Environment Protection and Biodiversity Conservation Act 1999*
GIS	Geographic Information System
HBOC	Hunter Bird Observers Club
IUCN	International Union for the Conservation of Nature
JAMBA	*Japan-Australia Migratory Bird Agreement*
KWRP	Kooragang Wetland Rehabilitation Project
NHT	Natural Heritage Trust
NPWS	National Parks and Wildlife Service (NSW)
NP	National Park
NR	Nature Reserve
NSW	Australian State of New South Wales
nMDS	non-metric Multi-Dimensional Scaling
MPS	Mean Particle Size
PRIMER	Plymouth Routines In Multivariate Ecological Research
QLD	Australian State of Queensland
Ramsar	*Convention on Wetlands 1971*
RIS	Ramsar Information Sheet
ROKAMBA	*Republic of Korea-Australia Migratory Bird Agreement*
SA	Australian State of South Australia
SE	Standard Error
SEPP14	State Environmental Planning Policy – No 14: Coastal Wetlands
SIMPER	Analysis of Similarity Percentages
SPSS	Statistical Package for the Social Sciences
TOC	Total Organic Carbon
TSC Act	*NSW Threatened Species Conservation Act 1995*
VIC	Australian State of Victoria
WA	Australian State of Western Australia
WWF	World Wide Fund for Nature

CHAPTER 1: GENERAL INTRODUCTION

1.1 Migratory shorebirds

1.1.1 Life history

Shorebirds, or waders, depend on coastal and inland wetlands and can occur in large numbers at their non-breeding and staging sites (Lane 1987; Barter 2002). Most migratory shorebirds are members of either the Scolopacidae (sandpipers, snipes, godwits, curlews and their allies) or Charadriidae (plovers and dotterels) families (Marchant & Higgins 1993; Higgins & Davies 1996). Their migration pathways, or flyways, represent specific routes used by multiple species. There are ten flyways around the world, including the East Pacific and West Atlantic flyways in the Americas, the East African and Mediterranean flyways which traverse Africa and Europe and the Indian Flyway through central and southern Asia (van de Kam *et al.* 2004). Australia is located in the East Asian-Australasian Flyway (EAAF) (Fig. 1.1) and is visited by 36 shorebird species that are regular migrants (Table 1.1) and an additional 16 species which are occasional vagrants from the other flyways (Watkins 1993) (see Appendix B). During the austral summer (Sept-Apr), a minimum of 4.8 million migratory shorebirds occur in Australia (Table 1.1), migrating each year from the northern hemisphere. In the EAAF, most migratory shorebird species breed either in Alaska, Siberia, Mongolia, Northern China or Japan during June and July of each year and migrate to Australia, South East Asia and New Zealand during their non-breeding season. The only exceptions are the Double-banded Plover *Charadrius bicinctus*, which breeds in New Zealand and spends its non-breeding season in Australia between February and September (Marchant & Higgins 1993), and the Australian Pratincole *Stiltia isabella*, which breeds in central and northern Australia and overwinters in Indonesia and Papua New Guinea between March and November (Higgins & Davies 1996).

During their large migrations, shorebird species have to withstand the high energy demands associated with continuous flying. Bar-tailed Godwits *Limosa lapponica baueri* can fly 10-11,000 km non-stop between their breeding grounds in western Alaska and non-breeding sites in New Zealand, the longest flight documented for a shorebird species (Gill *et al.* 2005; 2008). Smaller species, such as the Great Knot *Calidris tenuirostris* can also fly incredible distances (5,400 - 6,000 km) from north-western Australia to eastern China and Korea in around four days (Pennycuick & Battley 2003; Battley *et al.* 2004). Shorebird species undergo major physiological changes before their migration to fuel their long-distance flights (Landys-Ciannelli *et al.* 2003; Pennycuick &

Battley 2003; Battley & Piersma 2005). Many shorebirds increase the time they spend feeding before departure to increase their daily mass gains. Red Knots *Calidris canutus*, for example, gained on average 2.84 g per day at stop-over sites in the Wadden Sea, in the Netherlands (Nebel *et al.* 2000) and 2.93 g per day at stop-over sites in Iceland but daily mass gains increased steadily from 0.85 g to 7.0 g over a 24 day stop-over period (Piersma *et al.* 1999). A shorebird's preparation for migration is a two stage process. Flight muscles steadily increase in lean mass during the refuelling period and reach their maximum size before departure, while the lean mass of the main fuelling organs, the stomach, liver, kidney and intestines, undergo rapid growth in the early stages of the refuelling process to support flight muscle development (Piersma *et al.* 1999; Landys-Ciannelli *et al.* 2003). Immediately before departure, non-essential organs, such as the intestines, may be reduced in size (Landys-Ciannelli *et al.* 2003). After completing a successful migration, their arrival at the breeding grounds coincides with a brief arctic summer (June-July), where birds lay their eggs within one to two weeks and fledge their young six weeks later (Lane 1987).

1.1.2 Habitat requirements

During their migration, shorebird species use many wetland sites, spread across several countries, to sustain their energy supplies before they reach their destination on the breeding or non-breeding grounds. Most shorebird species that spend their non-breeding season in Australia rely heavily on coastal wetlands, while a few species will exploit inland wetlands when rainfall conditions allow (Thomas 1970; Kingsford & Porter 1993; Nebel *et al.* 2008). In coastal wetlands, shorebirds generally feed on invertebrates in exposed intertidal mudflats at low tide and then are forced to rest at high tide. High tide roosts are usually above the mean high water mark and are usually open, flat areas with a clear view, and an easy takeoff and landing (Lawler 1996). Although roosting requirements vary among species, most shorebirds select sites with low predation risks, low disturbance and a suitable microclimate, where their energy needs to remain thermo-neutral are minimal (Lawler 1996; Luis *et al.* 2001; Rogers 2003). In south-eastern Australia, typical roost sites include sand bars, ocean beaches, saltmarshes, artificial structures such as rock walls and oyster leases and occasionally mature mangroves (Lawler 1996).

Shorebirds generally feed at low tide and can be observed on beaches, intertidal mudflats, freshwater and brackish wetlands, farmland or saltmarshes (Lane 1987; Marchant & Higgins 1993; Higgins & Davies 1996) (Table 1.1). Most shorebirds prey on infaunal and epifaunal prey in the sediment (van de Kam *et al.* 2004) and consequently shorebird densities often match the

distribution of their preferred prey species (Goss-Custard 1970; Goss-Custard *et al.* 1977; Zharikov & Skilleter 2004c). In turn, prey density is often linked to sediment characteristics (Yates *et al.* 1999). Many shorebird species co-exist as they use different feeding methods, either visual or tactile, to locate their invertebrate prey (Dann 1987; Durell 2000). Small-scale differences in shorebird species distribution are usually related to their leg and bill length, as prey availability is determined by the depth to which a bird can wade and the maximum depth a bird can insert its bill into the sediment (Dann 1987). As a result, species are often segregated in their feeding habitats according to their preferences for water depth and sediment penetrability (Dann 1981b).

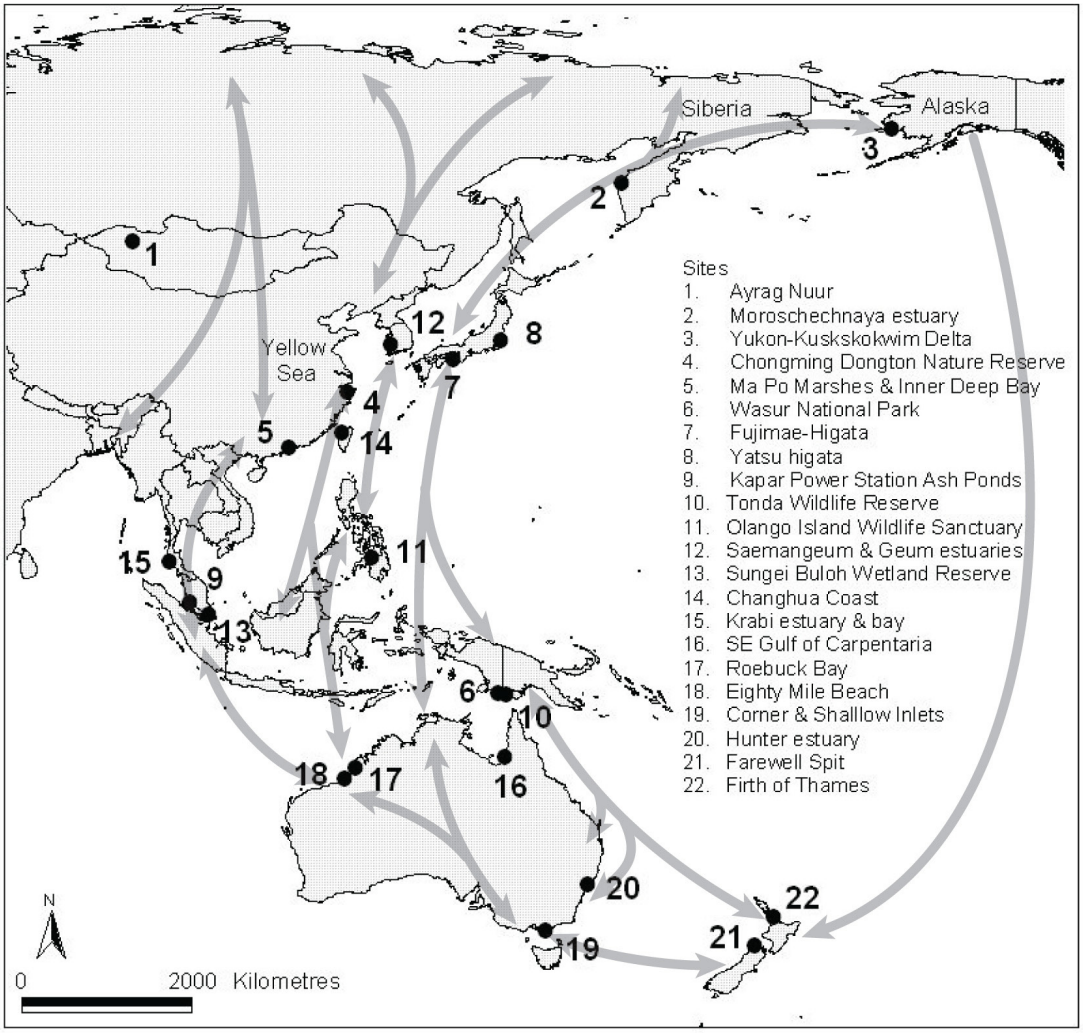


Figure 1.1 The East Asian-Australasian Flyway stretches from breeding sites in the Arctic to non-breeding sites in Australia and New Zealand. A selection of key shorebird sites in the EAAF are presented. In general, sites 1-3 are breeding sites, sites 4-15 are staging/ non-breeding sites and sites 16-22 are non-breeding sites (see section 1.2.2 for site descriptions).

Table 1.1 Conservation status, habitat use and population estimates of migratory shorebirds which spend their non-breeding season in Australia.

Common name[a]	Species	Breeding habitat[b]	Non-breeding habitat in Australia[b]	Population estimates[c]	
				Australian	Flyway
Asian Dowitcher (NT)	*Limnodromus semipalmatus*	Siberia, N China, Russia, Mongolia	Coastal wetlands, mudflats & sandflats	530	24 000
Bar-tailed Godwit (R)	*Limosa lapponica*	Northern Russia, Scandinavia, NW Alaska	Coastal, usually sheltered bays, estuaries with large intertidal flats	165 000	325 000
Black-tailed Godwit (V, R, NT)	*Limosa limosa*	Iceland, N Atlantic, Europe, Russia & China	Coastal, estuaries with large intertidal flats, occasionally inland	81 000	160 000
Broad-billed Sandpiper (V)	*Limicola falcinellus*	Scandinavia, Russia	Coastal wetlands, mudflats, estuaries	8 000	25 000
Common Greenshank	*Tringa nebularia*	Arctic circle, Siberia	Inland & coastal wetlands	20 000	60 000
Common Redshank	*Tringa tetanus*	Western Europe	Sheltered coastal wetlands	NA	75 000
Common Sandpiper (R)	*Actitis hypoleucos*	Western Europe, eastern Russia	Inland and coastal wetlands	4 500	25 000 – 100 000
Curlew Sandpiper	*Calidris ferruginea*	Arctic tundra	Intertidal mudflats, coastal lakes, estuaries, bays, occasionally inland	155 000	180 000
Double-banded Plover	*Charadrius bicinctus*	New Zealand	Estuarine & fresh or saline wetlands, grasslands & pasture	30 000	50 000
Eastern Curlew (r, E, v)	*Numenius madagascariensis*	Russia, NE China	Coastal wetlands, lagoons, intertidal areas & sandy spits	29 000	38 000
Great Knot (V, T, R)	*Calidris tenuirostris*	N Siberia	Coastal wetlands, intertidal mudflats, estuaries, lagoons & sandflats	360 000	375 000
Greater Sand Plover (V, R)	*Charadrius leschenaultii*	Siberia	Coastal wetlands, intertidal mudflats or sandflats, sandy beaches	74 000	110 000
Grey Plover	*Pluvialis squatarola*	Arctic tundra, Siberia, Alaska, Canada	Coastal wetlands, mudflats, sandflats, sandy beaches & rock platforms	12 000	125 000
Grey-tailed Tattler (T, R)	*Heteroscelus brevipes*	Siberia	Sheltered coasts with reef or rock platforms or intertidal mudflats	40 000	50 000
Latham's Snipe (R)	*Gallinago hardwickii*	Japan/ Siberia	Freshwater wetlands, inland, upland & coastal plains	15 000	36 000
Lesser Sand Plover (V, R)	*Charadrius mongolus*	Central & NE Asia	Coastal wetlands, estuaries, sandflats & mudflats	24 000	140 000
Little Curlew	*Numenius minutus*	Siberia	Coastal plains, grasslands, freshwater pools inland	180 000	180 000
Long-toed Stint (R)	*Calidris subminuta*	Siberia	Freshwater-brackish wetlands, muddy or vegetated shoreline	NA	25 000
Marsh Sandpiper	*Tringa stagnatilis*	Eastern Europe to Siberia	Coastal permanent or ephemeral wetlands, common inland	9 000	100 000 – 1 000 000
Oriental Plover	*Charadrius veredus*	Mongolia, E China	Inland-grasslands, beaches or wetlands	70 000	70 000

Table 1.1 (cont). Conservation status, habitat use and population estimates of migratory shorebirds which spend their non-breeding season in Australia.

Common name[a]	Species	Breeding habitat[b]	Non-breeding habitat in Australia[b]	Population estimates[c]	
				Australian	Flyway
Oriental Pratincole	*Glareola maldivarum*	China, India, Pakistan, Indonesia & Malay pens	Open country often near water, grassy flats & mudflats	2 880 000	2 880 000
Pacific Golden Plover (R)	*Pluvialis fulva*	N Siberia, Alaska	Mainly coastal, beaches, mudflats & sandflats	7 300	100 000 – 1 000 000
Pectoral Sandpiper	*Calidris melanotos*	N Russia, N America	Shallow fresh to saline wetlands, usually coastal but often inland	NA	NA
Pin-tailed Snipe	*Gallinago stenura*	Arctic tundra	Freshwater wetlands, grasslands	NA	25 000 – 1 000 000
Red Knot	*Calidris canutus*	N Siberia, Alaska	Intertidal mudflats, sandflats, estuaries & sandy beaches	135 000	220 000
Red-necked Stint	*Calidris ruficollis*	N. Siberia, Alaska	Coastal areas with mudflats, occasionally ocean beaches, common inland	245 000	325 000
Ruddy Turnstone (R)	*Arenaria interpres*	Northern Siberia, Alaska	Coastal areas with mudflats, rocky shores, rare inland	17 000	35 000
Ruff (R)	*Philomachus pugnax*	N Europe, Russia	Inland wetlands, brackish wetlands, with muddy edges, uncommon	NA	NA
Sanderling (V, R)	*Calidris alba*	High arctic in Russia, Greenland, Alaska	Open sandy beaches	8 000	22 000
Sharp-tailed Sandpiper	*Calidris acuminata*	NE Siberia	Muddy edges of fresh or brackish water, common intertidal & inland areas	140 000	160 000
Swinhoe's Snipe	*Gallinago megala*	Central Siberia, Mongolia	Freshwater wetlands, grasslands, uncommon but regular migrants to NW Australia	NA	25 000 – 100 000
Terek Sandpiper (V, T, R)	*Xenus cinereus*	Russia, eastern Europe	Intertidal coastal, mainly saline mudflats, lagoons & sandflats	25 000	60 000
Wandering Tattler	*Heteroscelus incanus*	Siberia, NW Canada	Rocky coasts, east coast & islands, not common in Australia	NA	NA
Whimbrel (R)	*Numenius phaeopus*	Siberia, Alaska	Intertidal coastal mudflats, river deltas & mangroves	10 000	100 000
Wood Sandpiper (R)	*Tringa glareola*	Scandinavia, N China, Siberia	Freshwater wetlands	6 000	100 000 – 1 000 000

[a]Regular non-breeding migrants are listed above but at least a further 16 species are occasional vagrants in Australia (see Appendix B).
Conservation status listed in parentheses: (NT) = near threatened under IUCN Red List criteria
(V) = vulnerable species under the New South Wales *Threatened Species Conservation Act 1995 (TSC Act)*
(v) = vulnerable and (R) = rare species under South Australia's *National Parks and Wildlife Act 1972*
(r) = rare species under the *Queensland Nature Conservation Act 1992*
(E) = endangered species under the Tasmanian *Threatened Species Protection Act 1995*
(T) = threatened species under the *Victorian Flora and Fauna Guarantee Act 1988*
[b]Breeding and non-breeding habitat descriptions taken from Marchant and Higgins (1993); Higgins and Davies (1996).
[c]Population estimates for Australia are based on DEH (2005) and estimates for the East Asian-Australasian Flyway (EAAF) are from Bamford *et al.* (2008). NA = not available.

1.2 Review of migratory shorebird conservation

1.2.1 Conservation mechanisms

The trans-equatorial lifestyle of migratory shorebird species challenges the effectiveness of many shorebird conservation programs. In the East Asian-Australasian Flyway, there are five international conventions, nine bilateral agreements and a flyway site network which offer protection for shorebird populations and/or their habitats (Table 1.2). The most widely known treaty is the Convention on Wetlands (Ramsar, Iran, 1971), which recognises internationally significant wetlands and, in particular, sites which provide waterbird habitat. The broad aim of the Convention is to stop the worldwide loss of wetlands and to conserve sites that remain through wise use and management. Australia has listed 65 Ramsar sites, covering more than 7.5 million hectares, since joining the convention as a contracting party in 1975 (Ramsar 2008). At a national level, the Australian Government addresses its obligations under the Ramsar Convention through the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act), however, it is the responsibility of the states and territories to nominate and manage each Ramsar site. Under the Ramsar Convention, developments that may affect the ecological character of a Ramsar wetland must be assessed by the Australian Government. There are also specific provisions under the *EPBC Act* for the protection and conservation of migratory shorebirds as a matter of national environmental significance. Further, the Wildlife Conservation Plan for Migratory Shorebirds (DEH 2006) was developed recently under the *EPBC Act* to direct research and management of migratory shorebird populations in Australia (Table 1.3).

Migratory shorebirds are also listed under bilateral agreements that Australia has with Japan (JAMBA, in 1981), China (CAMBA, in 1988) and the Republic of Korea (ROKAMBA, in 2007) (Table 1.2). Under these agreements signatory countries are obliged to protect migratory shorebirds and their roosting and feeding habitat. A further initiative, the East Asian-Australasian Shorebird Site Network (EAASSN), was set up in 1996 to identify, protect and manage important non-breeding and staging sites for migratory shorebirds in the EAAF. The shorebird site network is now managed under the Partnership for Conservation of Migratory Waterbirds and the Sustainable Use of their Habitats in the East Asian-Australasian Flyway (formerly the Asia-Pacific Migratory Waterbird Conservation Strategy and the Action Plan for the Conservation of Migratory Shorebirds in the East Asian-Australasian Flyway (2001-2005)) (Table 1.2). The new network incorporates significant sites for cranes, waterfowl and shorebirds. In total, 48 sites in 12

countries have been recognised as important sites for shorebirds under the network (Table 1.1). The listing of a site in the shorebird network is based on whether it regularly supports 20,000 shorebirds or 1% of an individual shorebird population. These action plans and initiatives are not legally binding but are designed to promote international cooperation between countries in the flyway and the ecological values of shorebirds and their habitats to local communities and decision-makers.

Despite Australia holding the record for the most Ramsar sites and shorebird network sites (17) in the flyway (Table 1.4), the ecological character of at least 22 of Australia's Ramsar sites has changed since their listing and overall 231 nationally important wetlands in Australia are under stress (Beeton *et al.* 2006). A lack of strategic conservation legislation limits Australia's capability to reduce the impact of threatening processes on shorebird habitats. For instance, upstream extraction has reduced flows into many significant wetlands in south-eastern Australia, including the Coorong, in South Australia (Fig. 1.1) (Phillips & Muller 2006) and the Macquarie Marshes, in New South Wales (NSW) (Kingsford 2000), which has had detrimental effects on waterbird numbers in recent years (Kingsford & Porter 2006; Paton & Rogers 2006; Porter *et al.* 2006; Nebel *et al.* 2008). This has occurred despite both areas being listed under the Ramsar Convention.

Some on-ground wetland conservation work in Australia is undertaken by non-government organisations such as the World Wide Fund for Nature (WWF) Australia, the Australasian Wader Study Group (AWSG), Birds Australia, the Australian Wetland Alliance (AWA), Conservation Volunteers Australia (CVA) and Wetlands International – Oceania. Many of these groups have received support from the Australian Government's Natural Heritage Trust (NHT) funding scheme to carry out shorebird conservation work. This includes the WWF's Shorebird Conservation Project which has organised field training and habitat management works including fencing, shelters and trails for public access at shorebird sites. The AWSG is a special interest group of Birds Australia which has coordinated shorebird monitoring programs and migration studies since the early 1980s (see section 1.4). Birds Australia, the AWSG and WWF-Australia recently set up the Shorebird 2020 program with funding from the Australian Government to coordinate and expand a national shorebird monitoring program (Clemens *et al.* 2008).

At the state level, for example, in NSW, there are at least eight legislative acts that aim to protect wetlands and/or shorebird species (Table 1.3). The NSW *Threatened Species and Conservation Act 1995 (TSC Act)* lists seven migratory shorebird species as vulnerable in NSW (Table 1.1) and coastal saltmarsh, in the NSW North Coast, Sydney Basin and South East Corner Bioregions, as an Endangered Ecological Community. Conservation reserves set up under the NSW *National Parks and Wildlife Act 1974*, *Fisheries Management Act 1994* and *Marine Parks Act 1997* provide some protection to shorebird habitats. National parks and reserves (*National Parks and Wildlife Act 1974*), however, only represent 3% of total wetland area in NSW (Kingsford *et al.* 2004), therefore, it is the responsibility of local governments and Catchment Management Authorities (CMAs) to manage remaining wetlands outside of the reserve system.

Overall, 51% of internationally important areas for shorebirds in Australia are not protected in the reserve system (Watkins 1993) and inland wetlands are generally under-represented (Smith 1991; Kingsford *et al.* 2004). The three most important shorebird sites in Australia, the south-east Gulf of Carpentaria (Queensland), Roebuck Bay and Eighty Mile Beach (Western Australia) (Fig. 1.1), support significant numbers of migratory shorebirds (up to 20 species) but these sites are not formally protected in the national reserve system (Watkins 1993). In a species assessment by Watkins (1993), important non-breeding sites for Great Knots and Black-tailed Godwits *Limosa limosa*, which are listed as threatened in several states (Table 1.1), were not protected within conservation reserves in Australia. Of the remaining sites which are partly/fully contained within reserve boundaries, many sites in south-eastern Australia are threatened by neighbouring land use changes and wider-scale catchment effects, including increased nutrient loads and water extraction (Gillanders & Kingsford 2002). For instance, the most significant shorebird site in NSW, the Hunter estuary (Fig. 1.1), is listed in both the Ramsar and flyway networks and a large proportion of its wetlands are reserved in the Hunter Wetlands National Park (DECC 2007). This non-breeding site, however, borders a large coastal port, the city of Newcastle which is the world's largest coal terminal, and inflows in the upper catchment are regulated for coal mining, power generation, irrigated agriculture and town water supplies. While shorebirds are a high priority in the management of the reserve (NPWS 1998a), insensitive catchment use and further port development are threats to the viability of remaining shorebird habitat in the estuary.

Table 1.2 International agreements and action plans that relate to migratory shorebird conservation in the East Asian-Australasian Flyway.

Agreement/ action plan	Date formalised	Abbreviation	Coverage		Details
			Wetlands	Shorebirds	
Convention on Wetlands	1975	Ramsar	Yes	Yes	Recognises & protects internationally important wetlands
Convention on Migratory Species (Bonn)	1983	CMS	Yes	Yes	Not specific to shorebirds, but aims to conserve avian migratory species throughout their range
Convention on International Trade in Endangered Species	1975	CITES	No	Yes	Protects species by monitoring & regulating international trade
World Heritage Convention	1975	WHC	Limited	Limited	Recognises internationally important cultural & natural heritage sites
Convention on Biological Diversity	1992	CBD	Limited	Limited	Dedicated to the conservation of biological diversity & sustainable use of its components
Important Bird Areas	2006	IBAs	Yes	Limited	Recognises sites that are important for globally threatened or restricted range species or sites that hold large number of species
Japan-Australia Migratory Bird Agreement	1981	JAMBA	Yes	Yes	Bilateral agreement for the protection of migratory birds common to Japan & Australia
China-Australia Migratory Bird Agreement	1988	CAMBA	Yes	Yes	Bilateral agreement for the protection of migratory birds common to China & Australia
Republic of Korea-Australia Migratory Bird Agreement	2007	ROKAMBA	Yes	Yes	Bilateral agreement for the protection of migratory birds common to the Republic of Korea & Australia
Asia-Pacific Migratory Waterbird Conservation Strategy	1996	APMWCS	No	Yes	Action plans & initiatives to promote the conservation of migratory waterbirds in the region
East Asian-Australasian Shorebird Site Network	1996	EAASSN	Yes	Yes	Identifies important wetlands for shorebirds & promotes conservation activities
Action Plan for the Conservation of Migratory Shorebirds in the East Asian-Australasian Flyway: 2001-2005	2001		No	Yes	Linked to Asia-Pacific Strategy, this action plan promotes shorebird conservation through international recognition of important sites in the East Asian-Australasian Flyway
The Partnership for the Conservation of Migratory Waterbirds & the Sustainable Use of their Habitats in the East Asian-Australasian Flyway	2006	Flyway Partnership	Yes	Yes	New international framework led by Governments of Australia & Japan & Wetlands International, to promote conservation of cranes, waterfowl & shorebirds & their habitats

Table 1.3 National and state (New South Wales) legislation and policy that relate to migratory shorebird conservation in Australia.

Level	Legislation/action plan	Date formalised	Abbreviation	Coverage		Details
				Wetlands	Shorebirds	
National	Environment Protection and Biodiversity Conservation Act	1999	EPBC	Yes	Yes	All migratory species & critical habitat are protected under Act as a matter of national significance
	Wildlife Conservation Plan for Migratory Shorebirds	2006		Yes	Yes	This plan sets out research & management actions needed to support species survival
State (NSW)	Threatened Species and Conservation Act	1995	TSC	Yes	Yes	Lists seven migratory shorebird species as vulnerable & coastal saltmarsh as a critically Endangered Ecological Community in NSW
	National Parks and Wildlife Act	1974		Yes	Yes	Protects native fauna and flora and this Act is the basis for the reserve system which protects areas containing outstanding or representative ecosystems
	Fisheries Management Act	1994		Yes	No	Prohibits damage to marine vegetation in NSW & provides for the declaration of aquatic reserves
	Marine Parks Act	1997		Yes	Yes	Basis for the creation of marine parks in NSW which includes sites with intertidal habitats important for shorebirds
	Wetlands Management Policy	1996		Yes	No	Main aim is to encourage the management of wetlands in NSW to halt the decline in wetland vegetation, biodiversity & water quality & to encourage Rehabilitation
	State Environment and Planning Policy No.14 - Coastal Wetlands (planning instrument under the Environmental Planning and Assessment Act 1979)	1985	SEPP 14	Yes	No	Main aim is to protect & preserve coastal wetlands in NSW & approval is required under Act before any clearing, draining, filling or levee construction is carried out on mapped coastal wetlands
	Coastal Protection Act	1979		Yes	No	Prohibits the development of the coastal zone without approval
	Protection of Environment Operations Act	1997	POEO	Yes	No	Prohibits negative changes to water quality
	Water Management Act	2000		Yes	No	Controls the extraction of water, construction of dams or weirs & any activities in or near water sources in NSW

1.2.2 Threats to shorebird populations in the East Asian-Australasian Flyway

Shorebird populations are threatened by habitat loss and the impacts of climate change throughout their flyways. Twenty percent of shorebird species that use the EAAF are listed as critically endangered or near threatened under IUCN risk criteria (Barter 2002). Within Australia, 18 species are listed under state or territory threatened species legislation and two species, the Asian Dowitcher *Limnodromus semipalmatus* and Black-tailed Godwit are globally threatened (Table 1.1). Overall, at least five species of migratory shorebirds have declined nationally and this has been attributed to the effects of habitat loss along the flyway (Olsen *et al.* 2003). As shorebird populations are closely linked to their food supplies, changes to their habitat can cause population declines. Global climate change projections suggest that sea level rise and prolonged droughts (Bates *et al.* 2008) may reduce the availability of coastal wetlands for migratory shorebirds (Galbraith *et al.* 2002; Austin & Rehfisch 2003), and that temperature increases may alter invertebrate prey reproduction (Lawrence & Soame 2004) and potentially cause a poleward shift in the range of many shorebird species (Chambers *et al.* 2005). Human-induced disturbance at high tide roost sites (Burton *et al.* 1996) and low tide feeding sites (Burger 1981; Thomas *et al.* 2003) can also result in higher energy expenditure and a reduction in food intake for birds at their non-breeding/staging sites (Stillman & Goss-Custard 2002; Coleman *et al.* 2003), which can impinge on their ability to build fat reserves to fulfil their annual cycle of moult, migration and breeding.

Threats to migratory shorebird populations vary in importance across the flyway. There are relatively few threats to migratory shorebird breeding habitat in the EAAF, compared to the main staging and non-breeding areas (Table 1.4). Shorebirds are distributed over large areas in their breeding range where human population density is generally low. Global climate change and the pressure to develop northwest Siberia and Arctic Alaska for new energy and mineral resources are the main threats to Arctic waterbird breeding habitat (Truett *et al.* 1997; Lindström & Agrell 1999; Zöckler & Lysenko 2000). The arrival of shorebirds on the breeding grounds in the Arctic region (Fig. 1.1) coincides with the snow-melt and an abundance of insect prey (Lane 1987; van de Kam *et al.* 2004). Changes in the timing of snow melt could reduce the reproductive success and survival of many migratory shorebird species, which may have difficulties adapting to relatively rapid climate change (Chambers *et al.* 2005).

In the EAAF, shorebirds share resources with 45% of the world's human population, which is concentrated in South Korea and China (Barter 2002). Many wetlands in south-east Asia are key staging sites in the flyway and are threatened by habitat loss and degradation (Table 1.4). The Yellow Sea (Fig.1.1) is the most important staging site for shorebirds in the flyway, supporting at least 60% of species and almost entire populations of at least six species that use this region during their migration (Barter 2002). Large areas of intertidal mudflats in the Yellow Sea have been reclaimed for agricultural, industrial and urban development. From 1917 to 1950, intertidal areas declined by 43% and 37% in South Korea and China respectively, and remaining habitats have been degraded by reduced river flows and high levels of pollution (Barter 2002). An important staging area in the Saemangeum estuary (South Korea), in the Yellow Sea, was destroyed after the construction of a large sea wall was completed in 2006 (Birds-Korea 2008). Formerly, this large area (40,000 ha) supported at least 192,000 shorebirds and 15 shorebird species in internationally significant numbers, including the globally endangered Spoonbill Sandpiper *Eurynorhynchus pygmeus* and the Nordmann's Greenshank *Tringa guttifer* (Moores *et al.* 2006). After the completion of the Saemangeum reclamation scheme, total numbers of shorebirds had declined by over 50% by 2007 (Moores *et al.* 2007) and further still in recent counts in 2008 (Moores *et al.* 2008).

Other staging sites along the flyway, e.g., the Chongming Dongtan Nature Reserve, China (Fig. 1.1), are close to major cities, and may be impacted by chemical, nutrient and sediment inputs from urban runoff (RIS 2001b) (Table 1.4). It is thought that the local harvesting of benthic invertebrates in South Korea and China may also impact on shorebird populations in the EAAF (Barter 2002). Over-harvesting of horse-shoe crabs has caused dramatic declines in Red Knot *Calidris canutus rufa* numbers at staging sites in the Americas (Baker *et al.* 2004; Morrison *et al.* 2004). Some shorebird species are also caught for food in parts of south-east Asia (Bamford 1992; Barter *et al.* 1997), including the Philippines (RIS 1994), Thailand (RIS 2001a) and Indonesia (RIS 2005). Furthermore, migratory shorebird species may be susceptible to outbreaks of highly pathogenic avian influenza viruses, which have been observed near some staging sites in the last decade (Tracey *et al.* 2004; Hurt *et al.* 2006).

Habitat loss and degradation is also the main threat to migratory shorebird species in their non-breeding range (Smith 1991; Watkins 1993; DEH 2005) (Table 1.4). Coastal development threatens many non-breeding sites in south-eastern Australia. Eighty-three percent of Australia's

19.4 million people live on the coast and 50% of estuaries have been modified by coastal developments (NLWRA 2002). The ecological effects of changes upstream of significant wetlands, such as vegetation clearing and river regulation, are also poorly understood (Bunn & Arthington 2002; Gillanders & Kingsford 2002). In NSW, for example, inflows into nine of its 17 inland catchments are heavily regulated for intensive agriculture (Kingsford *et al.* 2004) and major dams and diversions have impacted the health and extent of many significant inland wetlands (Kingsford 2000; Nebel *et al.* 2008). The hydrology of some coastal sites, such as the Hunter estuary, in NSW, has also been modified by industrial and urban development (Williams *et al.* 2000). Significant changes to the hydrology of the Hunter estuary have led to the encroachment of mangrove into saltmarsh habitats (Williams *et al.* 2000). This phenomenon is common in estuaries in south-eastern Australia, where 70% of sites have experienced saltmarsh losses, as a result of mangrove encroachment (Saintilan & Williams 1999). As shorebirds generally prefer open areas to roost and feed (Lawler 1996), the expansion of mangrove vegetation has reduced habitat quality for shorebirds in some areas of south-eastern Australia (Saintilan 2003; Straw & Saintilan 2005) and at the Firth of Thames Ramsar site (Fig.1.1) in New Zealand (Woodley 2005).

On their roosting and foraging grounds, shorebirds can suffer high disturbance rates by fishers, watercraft, walkers and dogs (Burger & Gochfeld 1991; Fitzpatrick & Bouchez 1998; Paton *et al.* 2000a; Blumstein *et al.* 2003) or coastal developments (Burton *et al.* 2002b; Durell *et al.* 2005). The loss of roost sites due to excessive disturbance and development has been linked to declines in shorebird populations in the United Kingdom (Mitchell *et al.* 1988) and the United States (Pfister *et al.* 1992). In Australia, roosting shorebirds are also vulnerable to predation by introduced predators, such as foxes and cats (Rogers *et al.* 2006a). Other introduced species can impact shorebird species indirectly. Exotic marine pests, for example, which can be introduced in ballast waters and by hull transport (Ruiz *et al.* 1997) could threaten intertidal benthic food sources along the Australian coast (Beeton *et al.* 2006). Saltmarsh and unvegetated mudflats have also been invaded by the introduced cordgrass *Spartina* spp. at some non-breeding sites in Australia, e.g., the Tamar estuary, in Tasmania (Adam 1981) and Western Port Bay, in Victoria (RIS 1999a). Originally introduced to reclaim land and stabilise mudflats (Laegdsgaard 2006), some studies have shown that hybrid *Spartina* can change invertebrate communities following colonisation (Hedge & Kriwoken 2000; Neira *et al.* 2006) causing shorebirds to avoid colonised mudflats (Goss-Custard & Moser 1988; Callaway & Josselyn 1992).

Table 1.4 Conservation status and threats to shorebird habitat at key breeding, staging and non-breeding sites in the East Asian-Australasian Flyway.

Range	Country	Ramsar sites [a]	Network sites [b]	Key sites	Province/ State	Area (ha)	Status [c]	Threats [d]
Breeding	Mongolia	11 (9)	0	Ayrag Nuur*	Hovd	45 000	R	3
	Russian Federation	35 (33)	1	Moroshechnaya Estuary	Kamchatka	219 000	CR, R, SN	3, 4
	Alaska	1 (1)	0	Yukon-Delta Nature Reserve	Alaska	7 689 027	CR	3, 4, 13
Staging	China	30 (16)	13	Chongming Dongtan Nature Reserve	Shanghai	100 000	CR, R, SN	1, 2, 3, 5, 8, 11, 12
				Mai Po Marshes & Inner Deep Bay	Hong Kong	1 540	CR, R, SN	1, 2, 3, 5
	Indonesia	3 (2)	1	Wasur National Park	Inan Jaya	413 180	CR, R, SN	1, 3, 6, 9, 10, 11
	Japan	33 (21)	8	Fujimae-Higata	Aichi Prefecture	323	CR, R, SN	1, 2, 3, 5
				Yatsu higata	Chiba Prefecture	40	CR, R, SN	2, 3, 5, 7
	Malaysia	5 (0)	1	Kapar Power Station Ash Ponds	Selangor	300	PM, SN	2, 3, 5
	Papua New Guinea	2 (1)	1	Tonda Wildlife Reserve	Western Province	590 000	CR, R, SN	3, 4, 8, 11
	Philippines	4 (2)	1	Olango Island Wildlife Sanctuary	Cebu	5 800	CR, R, SN	3, 5, 9
	Republic of Korea	8 (3)	1	Saemangeum & Geum estuaries	North Jeolla/ Chungham	40 100	-	1, 2, 3, 5, 8
	Singapore	0	1	Sungei Buloh Wetland Reserve	Kranji	87	CR, SN	1, 3, 5
	Taiwan	0	0	Tatu Estuary Wildlife Refuge	Taichung	2 670	-	1, 2, 3, 5, 14
	Thailand	10 (4)	1	Krabi Estuary & Bay	Krabi	21 299	CR, R, SN	2, 3, 5, 6, 7, 9, 12
Non-breeding	Australia	65 (25)	17	SE Gulf of Carpentaria	Queensland	NA	-	3, 4
				Roebuck Bay	Western Australia	55 000	R	2, 3, 4, 7, 10
				Eighty Mile Beach	Western Australia	125 000	R	2, 3, 4, 7, 10
				Corner Inlet	Victoria	67 186	CR, R, SN	3, 5, 7, 10, 11
				Hunter estuary	New South Wales	2 926	CR, R, SN	1, 2, 3, 5, 7, 10, 11
	New Zealand	6 (4)	2	Farewell Spit	South Island	11 388	CR, R, SN	3, 5, 10, 11. 12
				Firth of Thames	North Island	7 800	CR, R, SN	2, 3, 4, 5, 6, 10, 11

[a] Total number of Ramsar sites (and sites which are listed primarily for their importance to migratory waterbirds) as of 14 July 2008 (Ramsar 2008). Examples of key sites are provided for breeding, staging and non-breeding areas (see Figure 1.1 for site locations). * Note Ayrag Nuur is a breeding site for some shorebirds and staging site for other species.

[b] There are 48 sites in the flyway network as of July 2008 (Wetlands-International 2008).

[c] Conservation status: CR = Conservation Reserve, R = Ramsar site, SN = East Asian-Australasian Shorebird Network site, PM = Protected by management authorities.

[d] Main threats to sites and/or catchment: 1 = changes to hydrology (including dredging, artificial structures and water extraction); 2 = coastal development (residential/industrial); 3 = global climate change; 4 = resource extraction; 5 = declines in water quality (including urban runoff, sedimentation, oil spills); 6 = clearing of native vegetation; 7 = human-induced disturbance; 8 = overharvesting of benthic invertebrates; 9 = hunting; 10 = introduced animals; 11 = weeds/mangrove encroachment; 12 = aquaculture; 13 = fire; 14 = wind turbines. Sources: Information for site area and main threats was taken from Ramsar Information Sheets (Ramsar 2008) or provided by Wetlands International (Wetlands-International 2008). Information for sites not listed under the Ramsar convention was obtained from bird interest websites (Birding-Taiwan 2008; Birds-Korea 2008) (NA = not available).

1.3 Studying migratory shorebirds

Effective shorebird conservation is dependent on a detailed understanding of the distribution of shorebird populations, their life history and habitat requirements. There is a long history of research documenting shorebird population numbers and their behavioural ecology in western Europe (for reviews see: Evans *et al.* 1984; and van de Kam *et al.* 2004), but few research studies have focused on migratory shorebird species in Australia (see Table 1.5). Instead much of Australia's shorebird research has focused on population counts and migration studies primarily done by volunteer ornithologists (Straw *et al.* 2006).

1.3.1 Population counts

Long-term monitoring is essential for detecting changes in shorebird populations. Few studies monitor migratory shorebirds on their breeding sites, as their nests are widely spaced over remote areas (Meltofte 2001). In contrast, migratory shorebirds congregate in large numbers on their non-breeding and staging sites (Lane 1987; Barter 2002) making detailed surveys more feasible. Large-scale wildlife surveys have been undertaken since the early 1970s to monitor shorebirds in Europe and North America. In the United Kingdom, the 'Birds of Estuaries Inquiry' was established in 1970 by the British Trust for Ornithology, which involves synchronised monthly counts of multiple estuaries by amateur ornithologists (Goss-Custard & Moser 1988; Moser 1988). In North America, volunteers of the 'International Shorebird Survey' have surveyed non-breeding sites over each peak migration since 1972 (Howe & Geissler 1989). The Australasian Wader Study Group (AWSG) began annual summer and winter counts of shorebirds in 1981 to address a lack of information on shorebird numbers in Australia (Watkins 1993). In addition, shorebirds have been counted in annual aerial waterbird surveys of eastern Australia since 1983 (Kingsford *et al.* 1999; Nebel *et al.* 2008).

Ground or aerial-based surveys are often used to count shorebird species. Each method has its advantages and disadvantages and so a combination of survey methods is often adopted (Warnock *et al.* 1998; Kingsford 1999). Ground counts produce the most accurate estimates of shorebird numbers and are good for identifying cryptic or rare species. They are labour intensive but are generally inexpensive because they rely on volunteers. Surveys are simple so multiple counts can be carried out at many sites, and as ground counts are the most commonly used method, data are also comparable among sites. The main disadvantages of ground-based counts

are that they are not instantaneous, which increases the chances of double counting and they have limited coverage compared to wide-scale aerial surveys.

Aerial surveys cover large geographic areas and can be used to monitor large scale population trends over time. This method has been used to monitor shorebird (Garnett 1986; Kingsford & Ferster Levy 1997; Nebel *et al.* 2008) and waterbird populations (Braithwaite *et al.* 1986; Bayliss & Yeomans 1990; Kingsford *et al.* 1999) in many parts of Australia. Aerial surveys are often the best method for surveying remote locations and, although they provide imprecise instantaneous counts, these data are usually sufficient to demonstrate spatial and temporal changes in populations (Kingsford 1999). The main advantage of aerial surveys is that they are rapid and relatively inexpensive. Aerial waterbird surveys have been used in northern Australia (Garnett 1987; Morton *et al.* 1990, 1993; Chatto 2005), central Australia (Kingsford & Porter 1993, 1994) and in Papua New Guinea (Halse *et al.* 1996) to cover large areas of wetland which would be inaccessible on land (Kingsford & Porter 2009). Aerial surveys can also be used to locate marginal roost sites that may be overlooked in ground based counts. The main disadvantage of aerial surveys is that errors can arise through over- or under-estimates of flock sizes and cryptic or small species. In one study, where aerial and ground counts were compared directly, aerial surveys underestimated waterbird numbers by 50% for flock sizes greater than 40,000 birds, however, ground counts needed to cover the same area took 150 times longer and cost 14 times as much as the aerial surveys (Kingsford 1999). Colour bands and leg flags on shorebird species, used to estimate survival and determine migration routes, are also overlooked.

1.3.2 Migration studies

A number of techniques have been developed to investigate migratory behaviour in shorebird species. Regular counts are often used at key staging sites to determine dates of arrival and departure of migrating shorebirds (Battley 1997; Brayton & Schneider 2000; Nebel *et al.* 2000). Some studies also use radar to determine the direction and speed of departing shorebirds (Lane 1987). Banding or ringing studies are widely used and, although this method is labour intensive, re-sightings can provide important information on migratory pathways and habitat use. In Australia, the Commonwealth Scientific and Industrial Research Organisation (CSIRO) established the Australian Bird and Bat Banding Scheme (ABBBS) in 1953, but specific cannon-netting programs for banding shorebirds did not start till the early 1970s (Lane 1987). Colour banding has enabled the identification of critical habitats for many migratory shorebird species in

the EAAF (Weishu & Purchase 1987; Reigen 1999; Minton 2005). These studies have also provided information on longevity and site fidelity, while data taken at the point of capture (e.g., morphometrics, plumage condition and weight) have provided information on moulting strategies, age structures and breeding success (Barter 1988; Barter *et al.* 1988; Barter 1989b; Battley *et al.* 2004; Minton *et al.* 2005). The AWSG started a colour flagging program in 1990 which greatly increased the number of re-sightings of banded birds (Minton 2005). Individual colour flags are now available for 33 different locations within the EAAF. This program has been expanded so that individual birds can be identified from engraved colour flags with number/letter combinations. Engraved leg flags are currently being used to determine site usage and survival of godwits, knots and turnstones caught in New Zealand, Victoria and north-western Australia.

Recent technological developments have provided further information on shorebird migration strategies. Several studies have utilised stable isotope technology to track the wintering origins of migratory shorebirds in the American flyways (Farmer *et al.* 2003; Atkinson *et al.* 2005) or to identify the importance of different sites and habitat types (Hobson 1999). Recent developments in satellite tracking technology have revolutionised shorebird migration studies. Satellite technology was first trialled on shorebirds in Australia between 1997 and 1999, when custom-made harnesses were used to attach satellite transmitters to Eastern Curlews *Numenius madagascariensis* in Moreton Bay, Queensland. These birds were subsequently tracked to their breeding sites in north-eastern Russia (Driscoll & Ueta 2002). More recently, satellite transmitters were implanted in the abdominal cavity of Bar-tailed Godwits to track their migration between New Zealand and western Alaska (Gill 2006).

1.3.3 Behavioural studies

Instantaneous flock scans and focal observations of individual birds (Altmann 1974) are standard techniques in studies of shorebird behaviour. Focal animal sampling can be used to measure foraging and intake rates for individual birds (e.g., Goss-Custard 1977a; Goss-Custard *et al.* 1977; Stillman *et al.* 2000; Zharikov & Skilleter 2002), which can indicate habitat quality. Prey of large shorebird species can often be identified through observations of foraging birds. Experimental exclusion cages or prey manipulation can also be used to measure predation rates on invertebrate prey (Kent & Day 1983; Zharikov & Skilleter 2003a). Indirect measures of intake rates include the analysis of gut contents of shot or trapped birds (Reeder 1951; Poore *et al.* 1979;

Thomas 1986) or the analysis of faecal or regurgitated pellets (Mouritsen 1994; Scheiffarth 2001; Sitters *et al.* 2001; Zharikov & Skilleter 2004b) (Table 1.5). Digestion can make it difficult to accurately determine the contribution of each prey item from gut and faecal analysis. In addition to observational and diet-based studies, daily movements and time budgets of focal species can be tracked by marking individual birds (Burton *et al.* 2006) or tracking individuals. For example, radio-telemetry has helped to identify the location of nocturnal roost sites and the importance of nocturnal low tide feeding in studies of shorebird habitat use in Australia (Rohweder 2000; Richardson 2004; Rogers 2006) (Table 1.5).

Behavioural studies underpin a detailed understanding of the habitat requirements of shorebird species. Significant gaps in knowledge remain for many species that spend their non-breeding season in Australia, as only a limited number of studies have investigated shorebird roosting and foraging behaviour in Australia (Dann 1987) (Table 1.5). Given the scale of decline in coastal wetlands in south-eastern Australia, in particular, this lack of basic information limits effective management of shorebird habitats in their non-breeding range.

Table 1.5 Studies relating to migratory shorebird roosting and foraging ecology in Australia.

Type of study	Focal species	Study site	Description	Reference
Roosting	Multiple species	Shallow Inlet, VIC	Roost choice	(Jones 1985)
	Multiple species	Nine estuaries in NSW	Habitat models of roost choice	(Lawler 1996)*
	Little Curlew	Broome, WA	Nocturnal roosting ecology	(Jessop & Collins 1999)
	Multiple species	Richmond Estuary, NSW	Nocturnal roosts	(Rohweder 2001)
	Great Knot	Roebuck Bay, WA	Heat avoidance/migratory fuelling	(Battley 2002)*
	Great Knot	Roebuck Bay, WA	Heat avoidance in roosting birds	(Battley *et al.* 2003)
	Bar-tailed Godwit	Hunter Estuary, NSW	Radio-telemetry/habitat use	(Richardson 2004)*
	Bar-tailed Godwit	Hunter Estuary, NSW	Radio-telemetry/disturbance	(Foate 2005)*
	Great Knot, Red Knot	Roebuck Bay, WA	Habitat models of roost choice	(Rogers 2006)*
	Great Knot, Red Knot	Roebuck Bay, WA	Habitat models of roost choice	(Rogers *et al.* 2006a)
	Great Knot, Red Knot	Roebuck Bay, WA	Roost choice/disturbance	(Rogers *et al.* 2006b)
Foraging	Curlew Sandpiper, Red-necked Stint	South-eastern Tasmania	Feeding behaviour, gut contents	(Thomas & Dartnall 1971)
	Red-necked Stint, Sharp-tailed Sandpiper	Lake Reeve, VIC	Gut contents	(Poore *et al.* 1979)
	Red-necked Stint, Curlew Sandpiper, Sharp-tailed Sandpiper	Phillip Island, VIC	Feeding behaviour	(Dann 1981b)
	Multiple species	Lake Reeve, VIC	Feeding behaviour	(Dann 1983)
	Multiple species	Coorong, SA	Feeding behaviour, prey sampling & gut contents	(Paton 1984)
	Multiple species	South-eastern Tasmania	Gut contents	(Thomas 1986)
	Multiple species	South-eastern Tasmania	Pecking rates	(Thomas 1988)
	Great Knot, Red Knot	Broome, WA	Time budgets, prey types & invertebrate sampling	(Tulp & de Goeij 1991)
	Multiple species	Moreton Bay, Qld	Shorebird distribution & habitat use	(Thompson 1992)*
	Multiple species	Nine estuaries in NSW	Feeding habitat selection	(Lawler 1996)*
	Multiple species	Richmond Estuary, NSW	Day & night feeding behaviour	(Rohweder & Baverstock 1996)
	Multiple species	Moreton Bay, Qld	Shorebird distribution, sediment types	(Thompson 1998)
	Eastern Curlew	Western Port, VIC	Feeding behaviour	(Dann 1999c)
	Red-neck Stint, Curlew Sandpiper	Western Port, VIC	Feeding technique & gut contents	(Dann 1999a)
	Red-neck Stint, Curlew Sandpiper	Western Port, VIC	Feeding time, supra-tidal habitats	(Dann 1999b)
	Multiple species	Roebuck Bay, WA	Shorebird & prey density	(Rogers 1999)
	Multiple species	Rhyll, Phillip Island, VIC	Disturbance/flight distances	(Taylor & Bester 1999)
	Bar-tailed Godwits	Parramatta Estuary, NSW	Sex-related effects on intake rates/habitat use	(Taylor *et al.* 1999)

Table 1.5 (cont). Studies relating to migratory shorebird roosting and foraging ecology in Australia.

Type of study	Focal species	Study site	Description	Investigators
Foraging (cont)	Double-banded Plover	Western Port, VIC	Microhabitat use, feeding rates & prey choice	(Dann 2000a)
	Multiple species	Western Port, VIC	Feeding behaviour	(Dann 2000b)*
	Multiple species	Coorong, SA	Shorebird & prey density, sediment	(Paton *et al.* 2000b)
	Multiple species	Richmond Estuary, NSW	Day & night habitat use	(Rohweder 2000)*
	Latham's Snipe	Lower Hunter, NSW	Feeding rates, diet	(Todd 2000)
	Eastern Curlew	Moreton Bay, Qld	Low tide distribution	(Finn *et al.* 2001)
	Great Knot	Roebuck Bay, WA	Heat avoidance/ migratory fuelling	(Battley 2002)*
	Bar-tailed Godwit, Eastern Curlew	Moreton Bay, Qld	Sex & seasonal effects on intake rates	(Zharikov 2002)*
	Bar-tailed Godwit	Moreton Bay, Qld	Sex differences in habitat use/ intake rates	(Zharikov & Skilleter 2002)
	Great Knot	Roebuck Bay, WA	Heat avoidance in feeding birds	(Battley *et al.* 2003)
	Bar-tailed Godwit & other waterbirds	Botany Bay, NSW	Disturbance/flight distances	(Blumstein *et al.* 2003)
	Multiple species	North-western WA	Gut contents	(Jessop 2003)
	Bar-tailed Godwit	Moreton Bay, Qld	Prey depletion experiment	(Zharikov & Skilleter 2003a)
	Eastern Curlew	Moreton Bay, Qld	Intake rates, pellet & faecal analysis	(Zharikov & Skilleter 2003b)
	Bar-tailed Godwit	Hunter Estuary, NSW	Radio-telemetry study	(Richardson 2004)*
	Double-banded Plover	Richmond Estuary, NSW	Day & night feeding behaviour	(Rohweder & Lewis 2004)
	Eastern Curlew	Moreton Bay, Qld	Prey intake rates/migratory fuelling	(Zharikov & Skilleter 2004b)
	Eastern Curlew	Moreton Bay, Qld	Prey depletion experiment (bait harvesting)	(Zharikov & Skilleter 2004a)
	Eastern Curlew	Moreton Bay, Qld	Territory size, intake rates & prey density	(Zharikov & Skilleter 2004c)
	Eastern Curlew	Moreton Bay, Qld	Osmotic & digestive constraints, habitat use	(Blakey *et al.* 2006)
	Multiple species	Roebuck Bay, WA	Shorebird & prey density	(Rogers 2006)*
	Eastern Curlew	Moreton Bay, Qld	Feeding habitat selection	(Finn *et al.* 2007)

Note that this review excludes field notes and unpublished reports (*Research thesis).

1.4 Coastal wetlands

1.4.1 Wetland loss and degradation

In a global review, total wetland losses were estimated to be around 50% of their original area (Dugan 1993). The drainage of wetlands for agriculture is the principal cause of wetland losses in the northern hemisphere (Moser *et al.* 1996). In North America and Europe, up to 90% of floodplains have been cultivated for agriculture (Tockner & Stanford 2002). In the US, for example, rates of wetland loss were around 185,400 ha year^{-1} from the mid 1950s to 1970s but this declined to around 23,700 ha year^{-1} from 1986-97 (Dahl 2000). Furthermore, the UK has drained 23% of its estuaries and 50% of saltmarsh since Roman times (Davidson *et al.* 1991 from Moser *et al.* 1996). Large parts of Asia also have a long history of wetland reclamation, primarily for lowland rice farming. Other countries have altered their wetlands more recently. The Mesopotamian Marshes, in Iraq, for example, were once the most extensive wetlands in the Middle East, but major diversions and dam and channel construction upstream on the Euphrates and Tigris Rivers have devastated vast areas of wetlands in the Central Marshes (Scott 1995; Munro & Touron 1997).

Urbanisation is now the major cause of the loss and degradation of coastal wetlands in industrialised countries (Lee *et al.* 2006). In Australia, the condition of nationally important coastal wetlands varies greatly, with those in northern Australia in generally good condition and sites in southern Australia under most pressure. Urban development occupies over 25% and 15% of the NSW and Victoria coastlines respectively compared to just 7% in Queensland and around 1% of the Northern Territory's coastlines (Beeton *et al.* 2006). This has directly impacted coastal wetlands in the southern states: 17% of mangroves and 21% of saltmarshes in NSW and Victoria have been destroyed by coastal development (Turner *et al.* 2004). With the world coastal population predicted to approach six billion by 2025 (Kennish 2002), the impacts on coastal areas will inevitably increase and sea level rise predictions indicate that a further 22% of coastal wetlands could be lost worldwide by 2080 (Nicholls *et al.* 1999).

1.4.2 Rehabilitation and restoration

A recent shift in attitudes has led to an increased recognition of the value of wetlands (Streever *et al.* 1998). Coastal wetlands such as mangroves, seagrass beds, saltmarshes and intertidal mudflats, can be highly productive and act as wildlife corridors and nursery, breeding, feeding and resting grounds for many species of waterbirds (Lane 1987; Kingsford *et al.* 1999) and fish (Connolly 1994; Mazumder *et al.* 2006). They also provide essential ecological services, including water purification, nutrient retention, carbon sequestration, storm protection, flood mitigation, shoreline stabilisation and groundwater discharge (Mitsch & Gosselink 2000; Roy *et al.* 2001).

Many developers have now adopted habitat rehabilitation and creation to compensate for damaging natural habitats (Zedler 2004). Habitat restoration was conceived in North America twenty years ago in response to large scale wetland loss and has subsequently become popular in the UK, the Netherlands, Germany and Japan (Atkinson 2003; Pontee 2003). The principal goal of habitat rehabilitation is to return a damaged or degraded habitat to a fully functional ecosystem, irrespective of its original condition. Restoration is a special case of rehabilitation where the aim is to restore an ecosystem to as close to its original state as possible (Brown & Lugo 1994; Field 1998). Rehabilitation projects usually focus on the efficiency of the rehabilitation efforts in terms of labour and resource use, with the ecological success measured through assessments of biological diversity, water quality, productivity, nutrient retention and its vulnerability to invasions (Brown & Lugo 1994; Burchett *et al.* 1998; Field 1998).

Rehabilitation is complex and lengthy, usually creating habitats of lower quality than the habitats they replace (Wilkins *et al.* 2003; Zedler 2004). Waterbird communities are more diverse and abundant in natural wetlands compared to artificial ones (Melvin & Webb 1998; Havens *et al.* 2002; Atkinson *et al.* 2004; Darnell & Smith 2004; Ma *et al.* 2004). Furthermore, ongoing monitoring and rehabilitation works are often required (Evans *et al.* 1998; Neckles *et al.* 2002) as constructed wetlands take a long time to resemble reference wetlands (Brinson & Rheinhardt 1996; Havens *et al.* 2002). The socio-economic costs of rehabilitating and creating habitats can also be high. For example, the costs of rehabilitating coastal habitats ranged from US$ 2,000 - 160,000/ha for saltmarshes and US$ 3,000 - 510,000 ha^{-1} for mangroves (Spurgeon 1998).

In an intertidal setting, restoration and creation projects generally have a low success rate (Zedler & Callaway 2000), because wetlands are ecologically complex and very dynamic (Field 1998; Atkinson 2003). The main problem in the rehabilitation of waterbird habitat is a lack of detailed knowledge of the habitat requirements of target species (McKinstry & Anderson 2002). Constructed habitats need to function at a range of spatial scales for migratory shorebird species. Wetland connectivity being the most important factor for shorebirds at a landscape scale (Haig *et al.* 1998; McKinstry & Anderson 2002), while habitat diversity determines the abundance and diversity of migratory shorebirds at local scales (Atkinson *et al.* 2004).

1.5 Project background

1.5.1 The Hunter estuary

This study was done in the Hunter estuary in New South Wales (NSW), south-eastern Australia (32° 51'S/ 151° 46'E) from October 2004 to March 2007 (Fig. 1.2). The Hunter estuary lies at the end of the Hunter River which has a large catchment of around 21,425 km^2 (Kingsford *et al.* 2004). The lower estuary has been modified extensively to create the port of Newcastle (Kingsford & Ferster Levy 1997; Williams *et al.* 2000), which holds the world's largest coal export terminal. Extensive dredging; dyke, breakwater and ring canal construction; draining and land clearance for agriculture and industry; and the development of container shipping facilities and infrastructure have significantly altered estuarine habitats. This has consequently reduced the availability of roosting and feeding habitats for shorebird species (Geering & Winning 1993; Kingsford & Ferster Levy 1997).

The estuary now has two main river channels, a large tidal embayment, which receives semi-diurnal tides exposing about 750 ha of intertidal mudflat (Geering 1995) for 5.5 - 6.5 hours twice daily, and peripheral wetlands containing tidal creeks, mangrove, saltmarsh and freshwater wetland communities (Fig. 1.2). These habitats were gazetted as the Kooragang Nature Reserve in 1983, the largest estuarine reserve in NSW at the time (2,206 ha) (NPWS 1998a). The reserve was re-gazetted as the Hunter Wetlands National Park in 2007, to include Hexham Swamp Nature Reserve, and now covers about 4,255 ha (DECC 2007) (Fig. 1.2). Historically, the estuary supported 35 species of migratory shorebirds, including seven species of international (>1% flyway population) and 11 species of national (>1% Australian population) importance (see Chapter 2). It was ranked the fifth most important shorebird site in Australia in terms of species richness (Watkins 1993). Kooragang Nature Reserve was internationally recognised under the Ramsar Convention in 1984 (RIS 2002) and incorporated in the international East Asian-Australasian Shorebird Site Network in 1996. Total counts of shorebirds are thought to have declined over recent years from an estimated 7,900 in the 1970s to 5,300 in the 1980s and 3,700 in the 1990s (Kingsford & Ferster Levy 1997). Changes to the estuary and total shorebird numbers are discussed in more detail in Chapter 2.

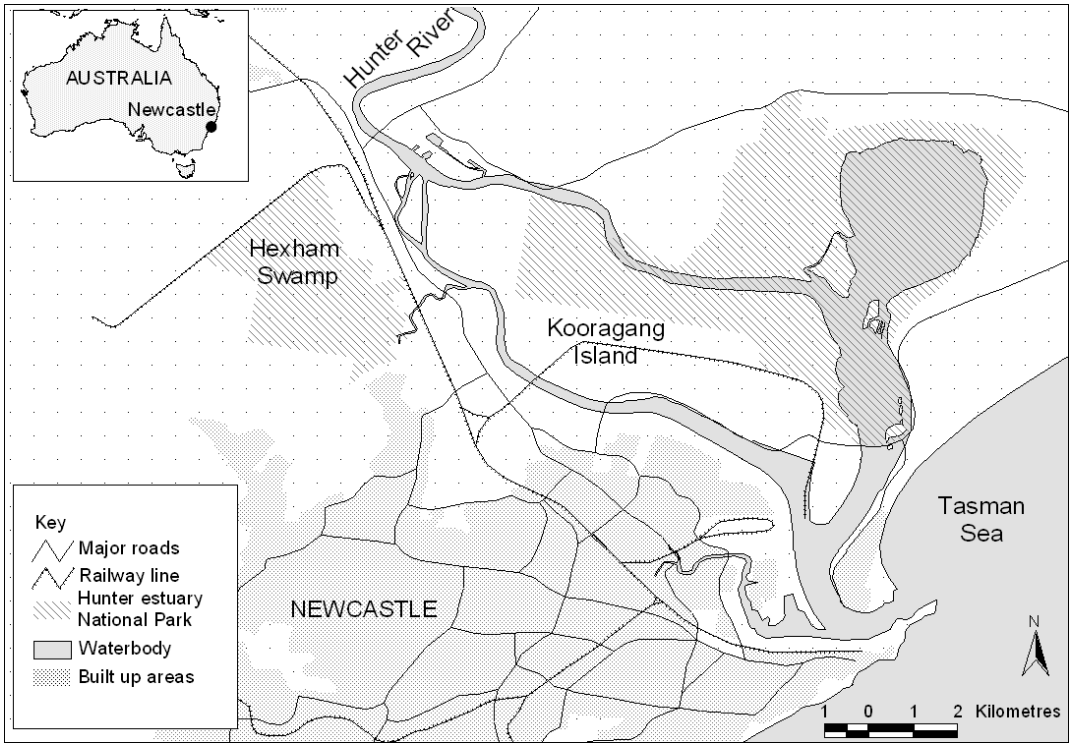


Figure 1.2 Location of the study site, the Hunter estuary, south-eastern Australia.

1.5.2 Project significance

This project forms part of a multi-disciplinary collaborative study between the Australian Catholic University and the University of Newcastle. The project was split into 'ecology' and 'ecohydraulics' PhD projects, with a combined aim of investigating the links between hydraulics, sediment, benthic invertebrates, vegetation and migratory shorebird habitat. As the ecology student, I investigated migratory shorebird ecology within the Hunter estuary. My study was complemented by an ecohydraulics study which investigated the hydraulic and vegetation characteristics of high tide roosts and the conditions that favour mangrove encroachment into saltmarsh in the Hunter estuary (Howe 2008).

The main aim of my study was to provide information on migratory shorebird habitat use at their most important non-breeding site in NSW, the Hunter estuary. The Commonwealth *EPBC Act* (1999) outlines the need for further information on the impacts of development on roosting and feeding habitats of migratory shorebirds, and how the secondary effects of wetland regulation, including altered hydrological regimes and mangrove encroachment, will influence the quality of

wetlands for shorebirds (DEH 2005). There are mechanisms for the Commonwealth to become involved in the approval of any actions likely to impact on migratory shorebird species and/or their habitats. This referral process cannot work effectively if the habitat requirements of shorebirds are poorly understood. In the Hunter estuary, high tide roost availability and the 'quality' of existing roosts is thought to be limiting migratory shorebird numbers (Geering 1995; Kingsford & Ferster Levy 1997). Nocturnal roost habitat has been most affected by development of the estuary, with the drainage of saltmarsh for agricultural land (Moss 1983) and encroachment of mangroves into areas of saltmarsh (Williams *et al.* 2000). Remaining roosting and feeding habitats in the Hunter estuary are now threatened by plans to expand the coal loading and port facilities in Newcastle, which will include further widening and deepening of the main shipping channels, and the construction of a third coal handling facility on Kooragang Island and new ship berths on the South Arm of the Hunter River (GHD 2003; NPC 2009).

The main outcomes of this study will be to provide information to assist the NSW Department of Environment and Climate Change (DECC) and the Kooragang Wetland Rehabilitation Project (KWRP) with the management of shorebird habitat in the Hunter estuary (Spencer & Howe 2008). Furthermore, the Hunter estuary was a case study from which general recommendations were made to assist management of shorebird habitat in other parts of south-eastern Australia.

1.6 Thesis Outline

In this first chapter, I introduced migratory shorebird ecology and their habitat requirements. I also reviewed current conservation mechanisms designed to protect migratory shorebird populations, threats to key shorebird sites along the EAAF, and techniques used to study shorebirds. The core chapters of this thesis (Chapters 2 to 7) were prepared as individual papers. Although there is some repetition in the description of the study site in the methods of each chapter, the bulk of the site description is located in Chapter 2.

Chapter 2 describes historical changes to estuarine habitats (1801-2007) and the abundance of migratory shorebirds (1965-2007) in the Hunter estuary. I also compared the magnitude of change in population trends in the Hunter estuary to two other non-breeding sites in south-eastern Australia: the Shoalhaven River estuary, in NSW and Corner Inlet, in Victoria (1981-2007). As this chapter was based on multiple data sets with different survey intensities and methodologies, I conducted a small experiment to investigate levels of observer error introduced through different count methods and observers (see Appendix A1).

In the remainder of my thesis, I focus on the roosting and foraging ecology of migratory shorebird species and the effectiveness of efforts to rehabilitate shorebird habitat in the Hunter estuary. The first step was to identify key habitats and the appropriate spatial scale of study given the high mobility of shorebird species. Four habitat measures were used as the main themes in my thesis: bird abundance; disturbance rates; foraging success; and prey availability.

In Chapter 3, I examined shorebird behaviour and microhabitat choice at the two major day roosts and one night roost. The protection of roost sites is important for shorebird species to rest free from disturbance. My aim was to determine how birds used the roost sites, the extent of disturbance and whether roost sites were limiting in the estuary. These questions were particularly relevant to shorebird nocturnal roosting ecology, given that nocturnal roosts have been lost from the Hunter estuary (Clarke & van Gessel 1983) and limited literature is available on the nocturnal behaviour of migratory shorebirds in Australia (Rohweder 2001; Rogers 2003) (Table 1.5). I focused on the Eastern Curlew *Numenius madagascariensis*, as it was a common species at roost sites in the Hunter estuary.

In the next three chapters, I focused on the foraging ecology of migratory shorebirds in the Hunter estuary, to identify the most important feeding habitats for shorebirds and the factors that determined their use. As only a limited number of studies have investigated the foraging behaviour of shorebirds in Australia (Dann 1987), these chapters also provide information that can be applied in a wider context for shorebird conservation in other parts of south-eastern Australia.

In Chapter 4, I investigated the behaviour of migratory shorebird species on artificial intertidal mudflats in the Hunter estuary. Mudflats artificially impounded by a river training wall were exposed for longer than the main low tide feeding habitat and therefore provided supplementary habitat for some shorebirds. I investigated the effect of tide type (spring or neap) and tidal period on the behaviour of migratory shorebird species in the largest mudflat impoundment in most detail. In Chapter 5, I measured the abundance of migratory shorebirds and their prey in six areas of intertidal mudflat within the Hunter estuary. To measure fine scale habitat use, I investigated the foraging behaviour of the Bar-tailed Godwit, which is common in the estuary. This species is gregarious and its movement can represent several other shorebird species within the Hunter estuary (Richardson 2004). In Chapter 6, I investigated the effects of the tidal period, time of day and microhabitat on the foraging behaviour of Sharp-tailed Sandpipers *Calidris acuminata* in saltmarsh. Saltmarsh was declared an Endangered Ecological Community in NSW in 2004 (*TSC Act 1995*) by a scientific determination that cited the importance of coastal saltmarsh as habitat for migratory shorebirds (DECC 2004), however, there are few studies that document migratory shorebird use of coastal saltmarsh in Australia (for a review see Spencer *et al.* 2009) (see Appendix A2).

In Chapter 7, I examined the effectiveness of attempts to rehabilitate migratory shorebird habitat in the Hunter estuary. This study was confined to a highly modified wetland on western Kooragang Island (Fig. 1.2), where tidal connectivity was reinstated by the removal of culverts in 1995. I compared low tide counts of migratory shorebirds collected during my study (2004-06) to data collected in a previous study done in 1994-97 (Kingsford *et al.* 1998). In the second part of this chapter I describe the results of a second experiment, where mangrove vegetation was cleared to restore shorebird habitat. These experiments followed a 'before and after control impact'

design (Underwood 1992), where shorebird numbers were recorded prior to and after the removal of culverts and mangrove vegetation to test the effectiveness of these rehabilitation techniques.

In the final chapter, Chapter 8, I summarise my main findings and discuss their implications for management of shorebird habitat in the Hunter estuary. I also suggest directions for further research that would form a logical progression from this study.

CHAPTER 2: CHANGES TO THE HUNTER ESTUARY, NEW SOUTH WALES, AUSTRALIA: IMPACTS ON MIGRATORY SHOREBIRD ABUNDANCE BETWEEN 1965 AND 2007

2.1 Abstract

In this chapter, I compared trends in migratory shorebird numbers in the Hunter estuary, New South Wales (NSW), to two other non-breeding sites in southeast Australia: the Shoalhaven River estuary (NSW); and Corner Inlet (Victoria) (1981-2007). I examined migratory shorebird population trends in the Hunter estuary in most detail from 1965-2007. There were significant declines (> 40%) in total numbers of migratory shorebirds at all three non-breeding sites (1981-2007). In the Hunter estuary, there has been a 42% decline in total numbers of migratory shorebirds since the 1980s and 28% decline since the 1990s. Based on maximum counts recorded from 2001-07, the Hunter estuary now only supports two species in internationally significant numbers; the Eastern Curlew *Numenius madagascariensis* (2% of flyway population) and Sharp-tailed Sandpiper *Calidris acuminata* (1% flyway population) compared to seven species listed from earlier records (1970-90). Trend analysis for the ten most common species, indicated that there were significant declines in four species but the power to detect significant trends for the remaining six species was generally poor. Declines were most severe (> 80%) for Curlew Sandpipers *Calidris ferruginea* but were also seen in Black-tailed Godwits *Limosa limosa* , Lesser Sand Plovers *Charadrius mongolus* and Pacific Golden Plovers *Pluvialis fulva.* The Hunter estuary is the most important site for shorebirds in the state of NSW, supporting up to 35 species during summer and 24 species during winter, but has been heavily modified since European settlement began in 1801. Major dredging and draining works from 1898-1928 and 1951-70 significantly altered the hydrological regime of the estuary. These engineering works formed the city of Newcastle, one of the world's most important coal ports, but also destroyed and degraded shorebird roosting and feeding habitat. The management of shorebird populations depends on the protection of habitats along their entire flyway. Declines in shorebird populations have occurred in the Hunter estuary, despite the site being protected in the national reserve system. Failure to adequately control ongoing port development and its impacts on estuarine habitats are contributing to declines and further declines may occur if developments proposed for the estuary and its upper catchment proceed.

2.2 Introduction

Coastal development has displaced shorebird species from many estuaries worldwide (e.g., Oosterschelde estuary, Netherlands (Schekkerman *et al.* 1994), Saemangeum estuary, South Korea (Moores *et al.* 2007), Cardiff Bay, Wales (Burton *et al.* 2002b), River Tees estuary, England (Evans *et al.* 1979), Tagus estuary, Portugal (Rosa *et al.* 2003)). This trend has also been observed in parts of eastern Australia (Gosbell & Clemens 2006; Nebel *et al.* 2008), where major ports and cities have developed along sheltered coastlines.

The Hunter estuary, in south-eastern Australia (Fig. 2.1) is home to the world's largest coal export terminal, but is also an important non-breeding site for migratory shorebird species (Smith 1991; Watkins 1993). Before European settlement, the Hunter River's floodplain was covered in rainforest and inhabited by the indigenous Awabakal, Worimi and Wanarua tribes (MHL 2003). When Europeans first arrived in the late 1700s, the Hunter estuary contained many islands separated by narrow intertidal channels, which would have provided extensive mudflats and sandflats (Fig. 2.1) for foraging shorebirds (Kingsford & Ferster Levy 1997). Early settlers used the estuary for fruit growing, dairy farming, timber harvesting and salt extraction (Kingsford & Ferster Levy 1997). Major dredging and draining works (1898-1928) reclaimed several islands in the main river channel (Williams *et al.* 2000) causing significant losses of shorebird roosting and feeding habitat (Table 2.1; Fig. 2.1). Levee banks were also installed around the perimeter of Fullerton Cove (Fig. 2.1) from 1913-28 to prevent flooding of adjacent agricultural land (Williams *et al.* 2000).

Between 1951 and 1989, a second stage of industrial development and expansion took place in the estuary. Inflows from the upper catchment were heavily regulated for irrigated agriculture, coal mining, power generation and domestic supplies, with major dams first constructed on the Hunter River in the 1950s. There are now 270 reservoirs in the Hunter River catchment, with a total storage capacity of around 1,670,000 ML (Kingsford *et al.* 2004). Construction of river training walls along the north and south arms of the Hunter River, after the *Newcastle Harbour Improvement Act* (1953) caused further losses of shorebird habitat (Coffey 1973; Kingsford & Ferster Levy 1997). The bulk of the dredging works were completed by 1967, after seven islands were amalgamated to form a single land mass, now known as Kooragang Island (Fig. 2.1), and an industrial railway line was completed to transport coal onto the newly reclaimed land (Table 2.1).

By the early 1970s, at least 704 ha of wetlands had been partially or fully reclaimed (NPWS 1998a) and the number of islands in the estuary reduced from 21 to six (Williams *et al.* 2000). These changes caused weed invasions, declines in water quality and reduced the total shoreline available as foraging habitat for shorebirds from 118 to 51 km (Kingsford & Ferster Levy 1997). Changes to the hydrological regime also altered the distribution of estuarine vegetation (Buckney 1987; Williams *et al.* 2000; MacDonald 2001). Mangroves expanded from 1310 ha in the mid 1950s to 1711 ha (31% increase) by the mid 1990s, largely replacing saltmarsh (declined from 2133 to 705 ha or 67%) but seagrass has not been seen in the Hunter River for more than three decades (Williams *et al.* 2000). Levee banks and ring drains around Fullerton Cove restricted tidal inundation further killing some mangroves (Moss 1983) and destroying saltmarsh, former night roosting habitat for shorebirds (Clarke & van Gessel 1983) (Table 2.1).

The natural values of the Hunter estuary and their degradation were first identified in the early 1970s (Coffey 1973; van Gessel & Kendall 1974; Dames & Moore 1978; Pressey & Middleton 1982; Moss 1983). A large portion of the estuary (2,206 ha) was listed as the Kooragang Nature Reserve in 1983 (NPWS 1998a) and later as a wetland of international importance under the Ramsar Convention (1984) (RIS 2002) and the East-Asian Australasian Shorebird Site Network (EAASSN) (1996). Rehabilitation began in the 1990s (see Table 2.1) with the Kooragang Wetland Rehabilitation Project (KWRP) established in 1993 to rehabilitate estuarine wetlands on Ash Island (780 ha), Tomago Island (800 ha) and Stockton sandspit (10 ha) (Fig. 2.2) (Svoboda 1996). Since the 1990s, the KWRP and NSW National Parks and Wildlife Service (NPWS) have continued to coordinate on-ground works in the estuary that create shorebird habitat (Table 2.1; see Chapter 7). In 2007, the Kooragang Nature Reserve was rezoned as the Hunter Wetlands National Park (4,255 ha) to include Hexham Swamp Nature Reserve (905 ha), originally listed in 1990, and an additional 423 ha on Tomago Island (DECC 2007) (Fig. 2.2).

The Hunter estuary's listing under these conservation reserves was largely in recognition of its importance to migratory shorebird species, most of which are listed under international treaties that Australia has with Japan, China and the Republic of Korea. Regionally, the Hunter estuary supported the greatest number of Black-tailed Godwits *Limosa limosa*, Terek Sandpipers *Xenus cinereus* and Broad-billed Sandpipers *Limicola falcinellus* in NSW (Smith 1991; Watkins 1993), which are listed as vulnerable under threatened species legislation (NSW *Threatened Species Conservation Act 1995*). The estuary is also known for supporting large concentrations

of more common species including: the Eastern Curlew *Numenius madagascariensis*, Bar-tailed Godwit *Limosa lapponica*, Curlew Sandpiper *Calidris ferruginea*, Common Greenshank *Tringa nebularia* and Pacific Golden Plover *Pluvialis fulva* (Coffey 1973; Smith 1991).

Remaining shorebird habitat is now threatened by a resurgence of development proposed for the Hunter estuary but estimates of the total number of shorebirds originally found within the estuary vary considerably. In light of major threats to estuarine habitats, it is critical that the current status of migratory shorebird populations in the estuary and the level of variability associated with counts of these species is determined. In this chapter, I investigated trends in migratory shorebird abundance in the Hunter estuary using counts collected between 1965 and 2007. I also assessed the significance of these population trends by comparing trends in the wider context of population changes at two other non-breeding sites in south-eastern Australia: the Shoalhaven estuary, NSW, and Corner Inlet, in Victoria (1981-2007).

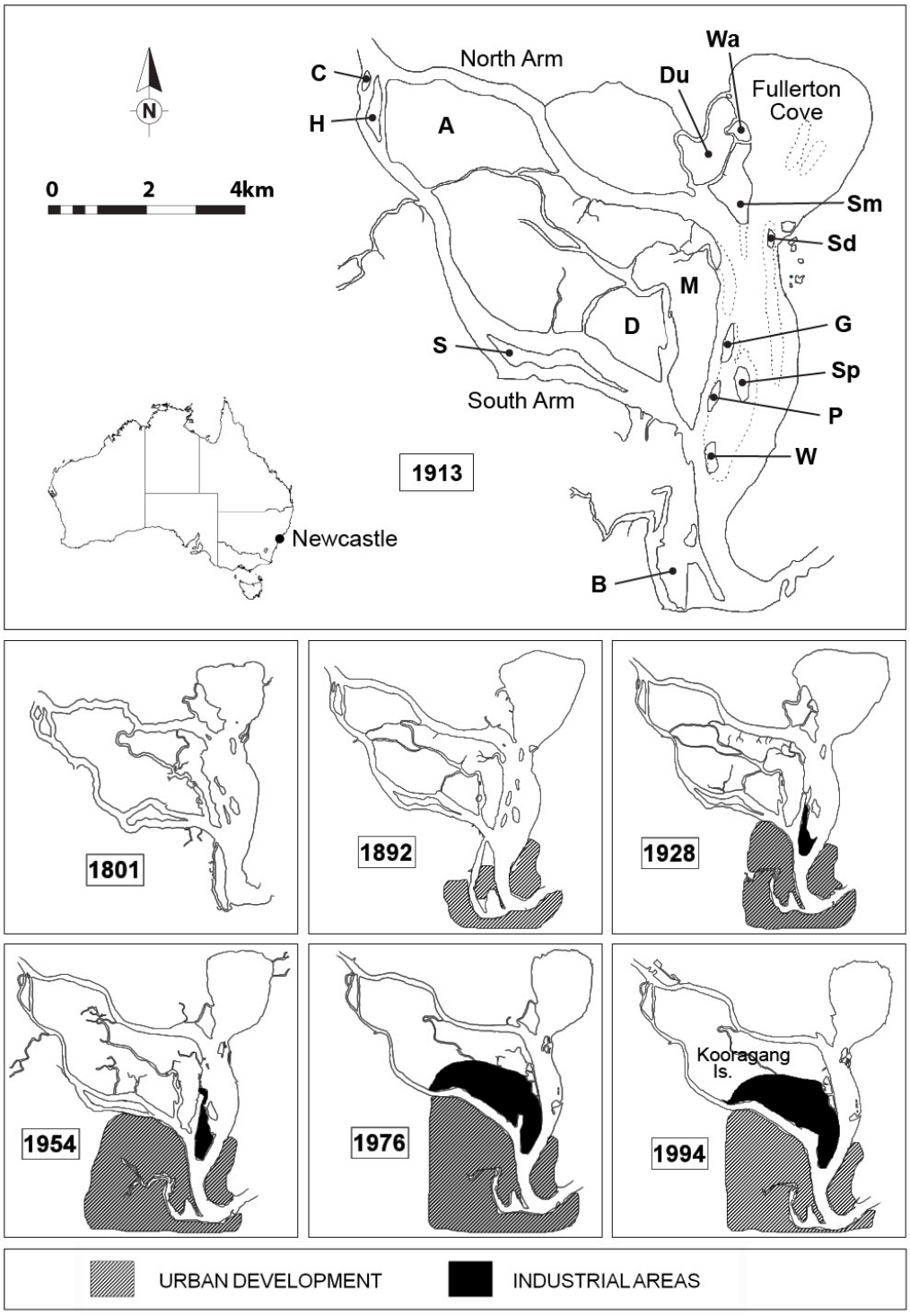


Figure 2.1 Major changes to the structure of the Hunter estuary from 1801-1994. The original islands were named as follows: C = Campbell, H = Hexham, A = Ash, D = Dempsey, S = Spit, M = Moscheto, B = Bullock, W = Walsh, P = Pig, Sp = Spectacle/Table, G = Goat, Sd = Sand, Sm = Smith, Du = Dunn's, Wa = Wallis (the dotted lines represent the original sand and mudflats) (adapted from Kingsford & Ferster Levy 1997).

Table 2.1 Major events in the Hunter estuary since European settlement and potential impacts on shorebird habitat (1801-2007).

Period	Description of events	Changes to shorebird habitat
1801-1844	Extraction of timber, coal & shell, construction of salt works, farming, fruit growing, construction of wharves & breakwater begin at the mouth of the Hunter River.	Loss of feeding habitat
1845-1892	Major floods in 1857, 1860 & 1893. Breakwater completed, major dredging of harbour begins, spoil dumped on islands, commercial prawn trawling begins, dairy farming starts on islands, copper works started, land reclaimed for Newcastle railway station & first islands merged following the construction of a training wall on the south arm of the Hunter River.	Loss of roosting & feeding habitat, disturbance of invertebrate prey
1893-1912	Construction of training wall on Walsh Island, sand flat removed from entrance to harbour & two islands removed from main river channel, Bullock Island merged with mainland.	Loss of roosting & feeding habitat, disturbance
1913-1928	Major steelworks established by 1915, more break walls built & wharves constructed, flood levee banks & radial drains built around Fullerton Cove, Walsh, Goat & Spectacle Islands reclaimed.	Major loss of feeding & roosting habitat, disturbance
1929-1941	Consolidation of islands & infilling of Kooragang Island tidal channels, large-scale removal of shell from Fullerton Cove for cement manufacturing, weir constructed between Ash & Hexham Islands.	Loss of roosting & feeding habitat, disturbance of major feeding habitat
1942-1954	Major dredging recommences in 1951, islands are further consolidated to create Kooragang Island by filling with dredged material.	Sand dump created temporary roosting habitat, loss & disturbance of feeding habitat
1955-1966	Major flood in 1955, Walsh Island joined to Moscheto Island after Moscheto Creek filled with dredged spoil, large river training wall (Kooragang dykes) is completed in 1966, construction of industrial railway line to transport coal onto Kooragang Island. Major dam construction upstream on the Hunter River for flood control.	Major loss & disturbance of roosting & feeding habitat, significant declines in flow & sediment inputs following dam construction may have impacted food supplies
1967-1976	Ash, Moscheto, Dempsey, Walsh, Spectacle, Pig & Goat Islands collectively named Kooragang Island in 1968. Completion of the industrial railway line & Stockton bridge, water pipeline, electricity powerlines & service roads installed on Kooragang Island, construction of floodgates restrict flows into Hexham Swamp, construction of a ring drain & heightening of levee banks around the perimeter of Fullerton Cove.	Loss of roosting habitat on Moscheto & Dempsey Islands, artificial roosting habitat created underneath Stockton bridge, degradation of feeding habitat at Hexham Swamp, loss of saltmarsh night roosts northwest of Fullerton Cove

Table 2.1 (cont). Major events in the Hunter estuary since European settlement and potential impacts on shorebird habitat (1801-2007).

Period	Description of events	Changes to shorebird habitat
1977-1992	Formal recognition of estuarine habitats following establishment of Kooragang Nature Reserve (NR) in 1983, Ramsar site in 1984 & declaration of SEPP 14 wetlands in 1985. Lostock Dam was constructed in 1983. Further deepening of entrance of Newcastle harbour, construction of gas pipeline at Ash Island, failure of culvert in 1990 increases flow onto Ash Island.	Kooragang dykes established as major day roost in lower estuary, major dam construction may have impacted shorebird food supplies
1992-2000	Establishment of the Kooragang Wetland Rehabilitation Project in 1993 & NR listed in shorebird flyway network in 1996. Culverts removed on two creeks to reinstate tidal flows on Ash Island in 1995, studies initiated to measure response of fish, waterbirds & wetland vegetation, lagoon created to provide day roosting habitat at Stockton sandspit but later this site was invaded by weeds & mangroves, artificial brackish wetland (Big Pond) is filled & water levels are raised to extend coal loading facilities.	Stockton sandspit provides day roosting habitat, but later degraded following mangrove encroachment, loss of supplementary feeding habitat for small shorebird species & night roost following the drainage of Big Pond
2001-2007	Stockton Sewage Treatment Works closed in 2002. Rehabilitation works undertaken to improve wetland habitat: mangroves removed from Stockton sandspit in 2002 & weir modified to improve tidal flushing of main lagoon, weed & mangrove sapling removal ongoing at Stockton sandspit, construction of five roosting platforms in NW Fullerton Cove, removal of mangroves & weeds from Smith & Sandy Islands in 2004, mangrove removal on Ash Island & reinstatement of tidal inundation into Hexham Swamp in 2007. Dredging works carried out along the south arm of the Hunter River in 2007. Declaration of the Hunter Wetlands National Park in July 2007.	Loss of roosting & feeding habitat for small shorebird species at Stockton Sewage works, successful rehabilitation of day roosting habitat at Stockton sandspit, potential rehabilitation of roosting & feeding habitat in other parts of the estuary

(Sources: Geering & Winning 1993; Kingsford & Ferster Levy 1997; Straw 1999; Williams *et al.* 2000). (See Figures 2.1 and 2.2 for site locations).

2.3 Methods

2.3.1 Site descriptions

The Hunter River is one of the longest coastal rivers in NSW draining a catchment of about 21,425 km^2 (Kingsford *et al.* 2004) (Fig. 2.2). The mouth of the Hunter River is a barrier estuary formed by the deposition of sediments in swamps and flats lying between inner and outer coastal barrier sands (West *et al.* 1985; Roy *et al.* 2001). The north and south arms of the river surround Kooragang Island (Fig. 2.2) (32° 51'S/ 151° 46'E), whose southern half is industrial land and northern and western portions are vegetated mainly by the mangrove *Avicennia marina* and saltmarsh species such as *Sarcocornia quinqueflora* and *Sporobolus virginicus* (Outhred & Buckney 1983; Winning 1996). Major day roosts for shorebirds, Stockton sandspit and the Kooragang dykes, are in the main channel while the main night roost, Windeyers Reach, is on Kooragang Island on the North Arm of the Hunter River (Fig. 2.2). Ash Island, on the western side of Kooragang Island (Fig. 2.2), contains saltmarsh that provides supplementary roosting and feeding habitat (see Chapter 7) (Geering 1995; Kingsford *et al.* 1998). Fullerton Cove, a shallow embayment on the north arm of the river, provides the most important feeding site, with 750 ha of intertidal mudflat exposed during spring low tides (Geering 1995). The Kooragang dykes also impound an additional 25 ha of intertidal mudflats, which provide supplementary feeding habitat for some species (see Chapter 4).

The Shoalhaven River estuary (34° 53'S/ 150° 44'E), NSW, is about 125 km south of Sydney and drains a catchment of 7,241 km^2 (Kingsford *et al.* 2004) (Fig. 2.2). Parts of the floodplain are used for dairy farming, while its upper reaches are dammed for water extraction. The mouths of the twin Shoalhaven and Crookhaven estuaries are partly protected (660 ha) under the Comerong Island Nature Reserve (NPWS 1998b). Intertidal mudflats in Comerong bay and Comerong lagoon provide feeding habitat for up to 27 species of migratory shorebirds (Kingsford 1990; Smith 1991; NPWS 1998b) (Fig. 2.2). This non-breeding site supports fewer migratory shorebirds than the Hunter estuary, but in the past total counts of Eastern Curlews have exceeded 1% of their flyway population and this site was also nationally significant for Pacific Golden Plovers, Lesser Sand Plovers *Charadrius mongolus* and Ruddy Turnstones *Arenaria interpres* (Smith 1991; Watkins 1993).

Corner Inlet (38 º 45'S/ 146º 18'E) in Victoria was ranked, with neighbouring Shallow Inlet (Fig. 2.2), as the fourth most important site for shorebirds in Australia in terms of species diversity (Watkins 1993). This non-breeding site is protected under the Corner Inlet Marine and Coastal Park, Corner Inlet Marine National Park and Wilson Promontory National Park (Fig. 2.2). Corner Inlet was also listed as a Ramsar site (67,186 ha) in 1982, which recognised its extensive tidal mudflats and large *Posidonia australis* seagrass beds (RIS 1999b). This site supports large numbers of resident and migratory shorebirds, including 50% of the Victorian migratory shorebird population during their non-breeding season (Martindale 1982). Counts of Curlew Sandpipers, Eastern Curlews and Red-necked Stints *Calidris ruficollis* have exceeded 1% of their flyway population estimates at this site (Watkins 1993).

2.3.2 Count data available for the Hunter estuary

Multiple sources of historical count data were available for shorebird populations in the Hunter estuary (Table 2.2). Shorebird species were first counted in the Hunter estuary in 1965 by the Newcastle Flora and Fauna Protection Society. Early publications list the regularity of each species' occurrence, maximum counts and rare sightings from 1965-81. Annual bird reports for NSW (NSW Field Ornithologists Club, Birding NSW), were available from 1971 onwards and included counts of shorebirds in the Hunter estuary and other sites in NSW. These reports summarise maximum counts of common species and rare occurrences of vagrant species for each year. The Hunter Bird Observers Club (HBOC) published annual reports for the Hunter region from 1993, while the Australasian Wader Study Group (AWSG) coordinated annual summer (Jan/Feb) and winter (Jun/Jul) high tide counts in the Hunter estuary (1982-2007), as a part of a national shorebird monitoring program. These annual counts are now coordinated by the HBOC which has done monthly estuary-wide high tide surveys since April 1999. In each survey month, multiple teams of observers count shorebirds at day roosts in the lower estuary and Ash Island simultaneously, using ground and boat-based surveys (see Appendix C). A total of 21 summer and 22 winter counts were completed in the Hunter estuary over the 26 year monitoring period (1982-2007). There were no annual counts in the summers of 1990, 1991, 1993, 1994 and 1998 or in the winters of 1991, 1993, 1997 and 1998.

Detailed studies of shorebird habitat use began in the Hunter estuary in 1992 (Table 2.2). Maximum counts were recorded for species during high tide counts of the Kooragang dykes over the 1992-93, 1993-94 and 1994-95 non-breeding seasons (Geering 1995). Counts of the most

abundant species were also recorded as they moved to feed in Fullerton Cove in 1993 and 1994 (Lawler 1996). The NSW NPWS monitored shorebirds in the Hunter estuary from March 1994 - May 1997, with ground surveys of waterbird habitat each month during summer (Sept - Mar) and bi-monthly in winter (Apr - Aug) (Kingsford & Ferster Levy 1997; Kingsford *et al.* 1998). Helicopter surveys of the whole estuary were also carried out bi-monthly from 1994-97, about 150 m from the shoreline at 100 ft (30.5 m) and a speed of 60 knots (111 km/hr). Shorebirds were grouped as either large or small species. At the same time, the Kooragang dykes and Ash Island were surveyed on the ground over three sequential days in each survey month (Kingsford & Ferster Levy 1997; Kingsford *et al.* 1998). Surveys of waterbird species were also done on Ash Island between 2001 and 2002 (Hutchinson & Morris 2003) and during this study (2004-07) (see Chapter 7 and Appendix E). Shorebird species were also counted during monthly high tide surveys of the main roosts from 1999 - 2003 (Straw 1999, 2000; Ekert 2003). The NPWS coordinated additional monthly high tide surveys of the lower estuary from October 2004 - March 2005. Boat and ground surveys of the roost sites were done simultaneously on three sequential days each month. At the same time as the NPWS surveys, I surveyed wetlands on Ash Island during high tide periods to determine the relative importance of these wetlands for migratory shorebird species (Table 2.2).

2.3.3 Other non-breeding sites

Annual counts were done over 27 years (1981-2007) in the Shoalhaven estuary (17 annual summer counts) and Corner Inlet (27 annual summer counts) as part of the AWSG national shorebird monitoring program (see section 2.3.2; Table 2.2) (AWSG database 1981-2007; Hewish 1992; Naismith 1992; Harris 1994b, a; Skewes 2002, 2003, 2004, 2005, 2007). No counts were available for the Shoalhaven estuary for the summers of 1991, 1994, 1996, 2000 and 2001. The NPWS Southern Directorate supplied count data for February 2006 (Craven *et al.* 2006).

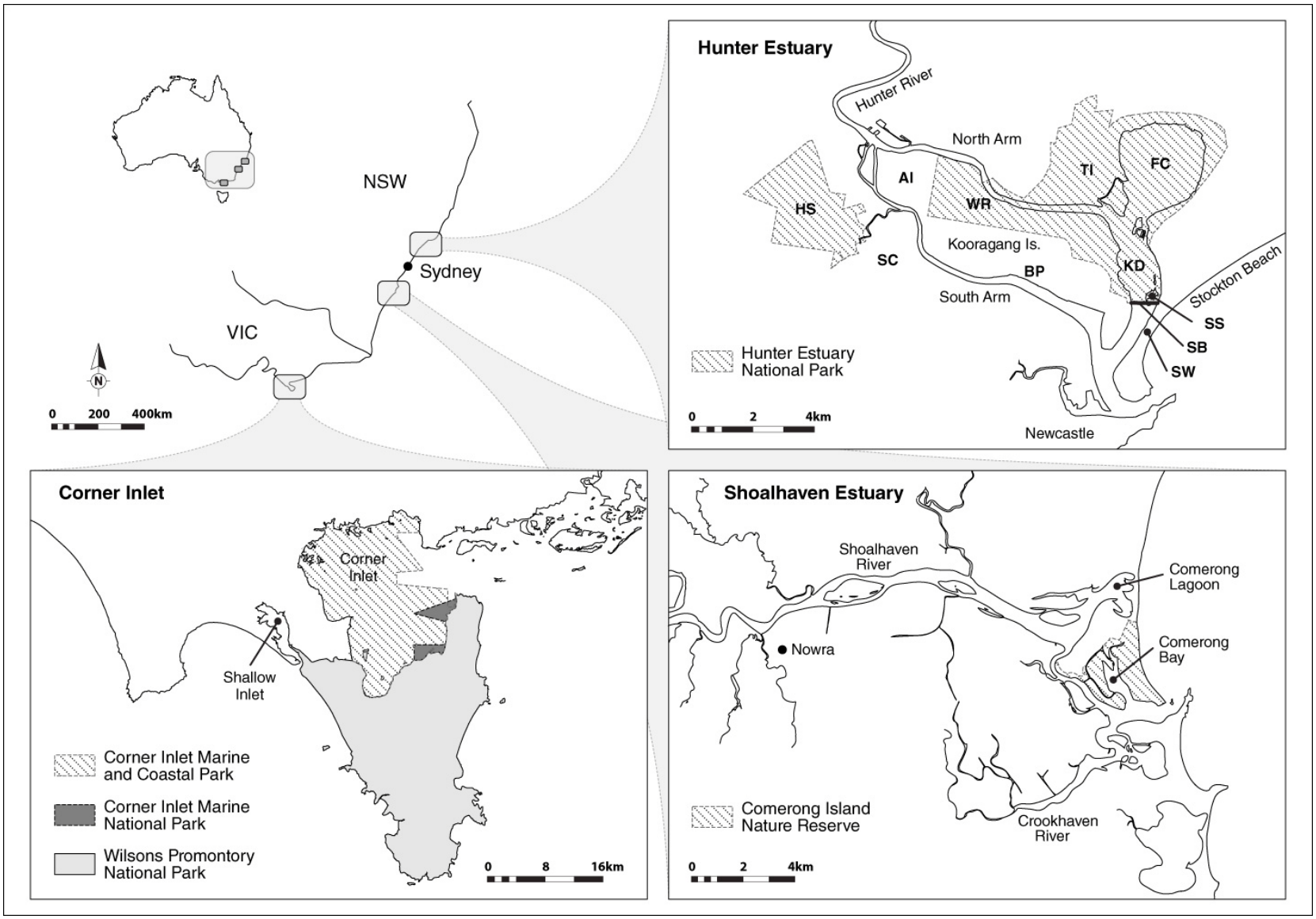


Figure 2.2 Location of the Hunter and Shoalhaven estuaries, in NSW, and Corner Inlet, in Victoria, south-eastern Australia.
(Hunter estuary site codes: AI = Ash Island, BP = Big Pond, FC = Fullerton Cove, HS = Hexham Swamp, KD = Kooragang Dykes, SB = Stockton Bridge, SC = Shortland Centre, SS = Stockton Sandspit, SW = Stockton Sewage Works, TI = Tomago Island, WR = Windeyers Reach).

Table 2.2 Availability of migratory shorebird count data for the Hunter estuary (1965-2007).

Dates	Description of data	Sources
1965-1981	*Ad-hoc* counts (summer & winter)	Early accounts by ornithologists (Holmes 1970; van Gessel *et al.* 1972; van Gessel & Kendall 1972a, b; van Gessel 1973; Gosper 1974; van Gessel & Kendall 1974; Morris 1975; van Gessel 1976; HBOC 1979; Gosper 1981)
1972-2002	*Ad-hoc* observations, maximum counts & rare sightings (summer & winter)	NSW annual bird reports (Rogers 1972, 1973, 1974, 1975, 1976, 1977; Rogers & Lindsey 1978; Lindsey 1979; Lindsey 1980, 1981, 1982, 1984, 1985, 1986; Coopers 1989, 1990, 1991; Morris & Burton 1992; Burton & Morris 1993; Morris & Burton 1993, 1994, 1995, 1996, 1997, 1999; Morris 2000, 2001, 2002b, a, 2003, 2004)
1982-1999	Annual summer & winter counts	AWSG national monitoring program (AWSG database 1982-94; Hewish 1986; Hewish 1987a, b, 1988, 1989b, a, 1990a, b; Harris 1995a, b, 1996, 1997, 1999; Harris 2000)
1983-2007	Population estimates for the Hunter region	Population status reports (Morris 1983; Smith 1991; Watkins 1993; Herbert 2007; Stuart *in prep.*)
1992-2007	Studies of shorebird habitat use	Research projects (Geering & Winning 1993; Geering 1995; Lawler 1996; Kingsford & Ferster Levy 1997; Kingsford *et al.* 1998; Hutchinson & Morris 2003; Richardson 2004; Foate 2005; Crawford & Herbert 2007; J. Spencer *pers. obs.*)
1993-2007	*Ad-hoc* observations & counts from estuary-wide monthly surveys of high tide roosts (1999-ongoing)	HBOC bird reports (Stuart 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008)
1999-2003	High tide counts of main roosts	Monitoring projects (Straw 1999, 2000; Ekert 2003)
2004-2005	Estuary-wide monthly counts of high tide roosts	Surveys coordinated by NSW NPWS (Hunter region) alongside surveys of Ash Island (J. Spencer *pers. obs.*)

2.3.4 Historical count data and statistical analyses

Most migratory shorebirds arrive in Australia in late August/ early September and depart by the end of April (Lane 1987). I summarised available counts for the Hunter estuary from 1965-2007 (primary sources) for summer (late August - April) and winter months (May - early August). I did not include counts collated in population status reports for the Hunter estuary by Smith (1991) or Watkins (1993). The Spotted Redshank *Tringa erythropus* and Oriental Pratincole *Glareola maldivarum* were not included as records for both species were only listed in the compiled reports of Smith (1991) and unconfirmed records for the Cox's Sandpiper *Calidris paramelanotos* (Coopers 1992) were also excluded. A maximum count of 520 Ruddy Turnstones for the Hunter estuary was recorded in both Smith (1991) and Watkins (1993) but this record was an error in data transcription from the original AWSG surveys in February 1986. Furthermore, two records of 3,000 Black-tailed Godwits recorded in the 1984 and 1985 NSW bird reports (Lindsey 1986; Coopers 1989) were also excluded from the analysis, as these counts were not consistent with two AWSG counts of 520 and 800 Black-tailed Godwits recorded in February 1984 and 1985 respectively, and were most likely misidentified Bar-tailed Godwits. I excluded records from Hexham Swamp, Shortland Wetlands and Stockton Beach (Fig. 2.2) as these sites were not surveyed regularly.

Only maximum counts of shorebird species were available from the annual NSW bird reports and other early publications for the period 1965-81 (Table 2.2). Records from the annual bird reports generally refer to the Hunter estuary as a whole with no details of the locality within the estuary or tide state at the time of sighting, while counts recorded during the AWSG and HBOC surveys (from 1982 onwards) were conducted at high tide at roost sites in the estuary. From 1982, mean counts for five year periods were calculated for each species using the AWSG/ HBOC annual summer (Jan/ Feb) and winter (Jun/ July) surveys, where data were available. As large flocks can pass through the estuary, maximum counts were also collated from other studies available (see Table 2.2). I also calculated means and standard errors from estuary-wide counts done by the NPWS in February from 1995 - 97 (n = 8), in July from 1994 - 96 (n = 8) (Kingsford & Ferster Levy 1997; Kingsford *et al.* 1998) and in February 2005 (n = 3) (Table 2.2). I examined all survey months for each summer or winter period to determine maximum counts for both monitoring programs (Table 2.3).

To analyse long-term changes in shorebird numbers in the Hunter estuary, I selected ten species to investigate individual population trends: Bar-tailed Godwit, Black-tailed Godwit, Common

Greenshank, Curlew Sandpiper, Eastern Curlew, Grey-tailed Tattler *Heteroscelus brevipes*, Lesser Sand Plover, Marsh Sandpiper *Tringa stagnatilis*, Pacific Golden Plover and Terek Sandpiper. I selected species that were historically common in the estuary in summer months, excluding passage migrants, highly mobile, cryptic, solitary or rare species whose abundance can be difficult to interpret (Prys-Jones *et al.* 1994). Red-necked Stints and Red Knots *Calidris canutus* can occur in large numbers (between September and October) before moving to sites further south of the Hunter estuary and are only found in small numbers in other summer months (Gosper 1981). Sharp-tailed Sandpipers *Calidris acuminata* distribution varies widely with inland or coastal areas wetland availability in Australia (Higgins & Davies 1996) making trend detection difficult. Cryptic species (e.g., Common Sandpiper *Actitis hypoleucos*, Latham's Snipe *Gallinago hardwickii*) are regular migrants but are generally only observed after disturbance (Higgins & Davies 1996). Rare species (e.g., Pectoral Sandpiper *Calidris melanotos*, Long-toed Stint *Calidris subminuta*), not found in numbers greater than 1% of their flyway population (Watkins 1993), may also be underestimated as their detection is reliant on the presence of experienced observers.

I used simple linear regressions to investigate population trends with analysis of variance (SPSS 2005) to determine the significance of the least square regression lines (Fowler *et al.* 1998). Total counts for all trend analyses were log-transformed to meet assumptions of normality and homogeneity in the variances (Quinn & Keough 2002). Two-tailed Durbin-Watson tests were used to test for autocorrelation in the residuals. I compared trends in total numbers of migratory shorebirds in the Hunter estuary to counts from the Shoalhaven estuary and Corner Inlet from 1981 - 2007. To compare among sites, I used the annual summer counts (Jan/Feb) recorded by the AWSG/ HBOC in the Hunter estuary (1982 -2007; n = 21), the AWSG and NPWS Southern Directorate in the Shoalhaven estuary (1986 - 2007; n = 17) and the AWSG in Corner Inlet (1981 - 2007; n = 27) when numbers of migratory shorebirds are relatively stable before birds begin their northward migration in late March and early April (Lane 1987). I also used linear regression analyses to determine if mean winter (Jun/Jul) counts (1982 - 2007; n = 22) and maximum summer (Sept-Apr) and winter (May-Aug) counts (1965-2007) differed significantly among survey years for the Hunter estuary. Maximum counts were available for seven survey periods which covered two or more years (Tables 2.3; 2.4). I analysed total summer (Jan/Feb) counts of six species for the site comparisons. Black-tailed Godwits, Grey-tailed Tattlers, Marsh Sandpipers and Terek Sandpipers were excluded from the site comparison as these species are not common south of the Hunter estuary (Lane 1987; Watkins 1993). Also, Curlew Sandpipers were not in sufficient numbers in the Shoalhaven estuary to be

analysed while Pacific Golden Plovers and Lesser Sand Plovers were uncommon at Corner Inlet. I also used linear regression analyses to determine whether mean counts of the ten selected species differed across years in the Hunter estuary. This analysis was based on single summer (Jan/Feb) counts from 1982-99 but from 2000 onwards, mean counts and confidence intervals were calculated for the core non-breeding season (Oct-Mar) using HBOC data. March counts of Curlew Sandpipers were excluded as counts of this species were considerably lower in this month, indicating an early departure date. I used a *post hoc* power analysis (Hintze 2008) to calculate the power of non-significant trends for species in the Hunter estuary.

2.4 Results

2.4.1 Total numbers of migratory shorebirds

There were significant long-term declines in numbers of migratory shorebirds at the non-breeding sites (1981-2007), despite large variation in annual summer counts which produced poorly fitted regression lines (Fig. 2.3). Total numbers of migratory shorebirds declined by over 40% at the three sites, a decline of around 1.5% per year. In Corner Inlet, total summer counts declined from 6,406 (± 937 S.E.) in the 1980s, 5,155 (± 738 S.E.) in the 1990s to about 3,806 (± 921 S.E.) birds in 2000-07, despite a large increase in birds in 2007 (Fig. 2.3). Total counts in the Shoalhaven estuary were only 787 (± 133 S.E.) in 2000-07 and 895 (± 151 S.E.) in the 1990s compared to 1,504 (± 257 S.E.) birds in the 1980s. Mean summer counts in the Hunter estuary declined from 5,264 (± 480 S.E.) in the 1980s, to 4,244 (± 778 S.E.) in the 1990s and 3,036 (± 130 S.E.) birds in 2000-07 (Fig. 2.3; Fig. 2.4). There was also a 61% decline in the mean number of shorebirds remaining in the Hunter estuary over winter months: 925 (± 116 S.E.) birds in the 1980s, 450 (± 68 S.E.) in the 1990s and 358 (± 34 S.E.) birds in 2000-07 ($r^2 = 0.77$, $F_{1,20} = 29.1$, $p < 0.001$). Maximum counts of migratory shorebirds in the Hunter estuary did not decline over summer ($r^2 = 0.38$, $F_{1,5} = 3.0$, $p = 0.142$) or winter ($r^2 = 0.39$, $F_{1,5} = 3.2$, $p = 0.132$) (1965 - 2007) (Tables 2.3; 2.4).

2.4.2 Species diversity in the Hunter estuary

Overall, 35 species of migratory shorebirds were recorded in the Hunter estuary in summer and 24 species in winter months (1965 - 2007). The total number of species was similar in each decade (26 - 30 species) only varying according to sightings of vagrant and rare species (Tables 2.3; 2.4). Five vagrant species (Buff-breasted Sandpiper *Tryngites subruficollis*, Hudsonian Godwit *Limosa haemastica*, Lesser Yellowlegs *Tringa flavipes*, Little Stint *Calidris minuta* and Ringed Plover *Charadrius dubius*) were observed in the Hunter estuary in summer. Five migrant species which occur in numbers < 1% of their flyway populations in Australia (Asian Dowitcher *Limnodromus semipalmatus*, Long-toed Stint, Pectoral Sandpiper, Ruff *Calidris pugnax* and Wandering Tattler *Tringa incana*) were also observed occasionally. Another seven species in low numbers in NSW also occurred in the estuary: the Broad-billed Sandpiper, Greater Sand Plover *Charadrius leschenaultii*, Grey Plover *Pluvialis squatarola*, Little Curlew *Numenius minutus*, Oriental Plover *Charadrius veredus*, Sanderling *Calidris alba* and Wood Sandpiper *Tringa glareola* (Tables 2.3; 2.4). Of the more common species (18 in total) in summer months, Bar-tailed Godwits, Curlew Sandpipers and Eastern Curlews were observed in largest numbers (Table 2.3). Large flocks (>200

birds) of Bar-tailed Godwits and Eastern Curlews also remained in the Hunter estuary over winter (May-Aug) (Table 2.4).

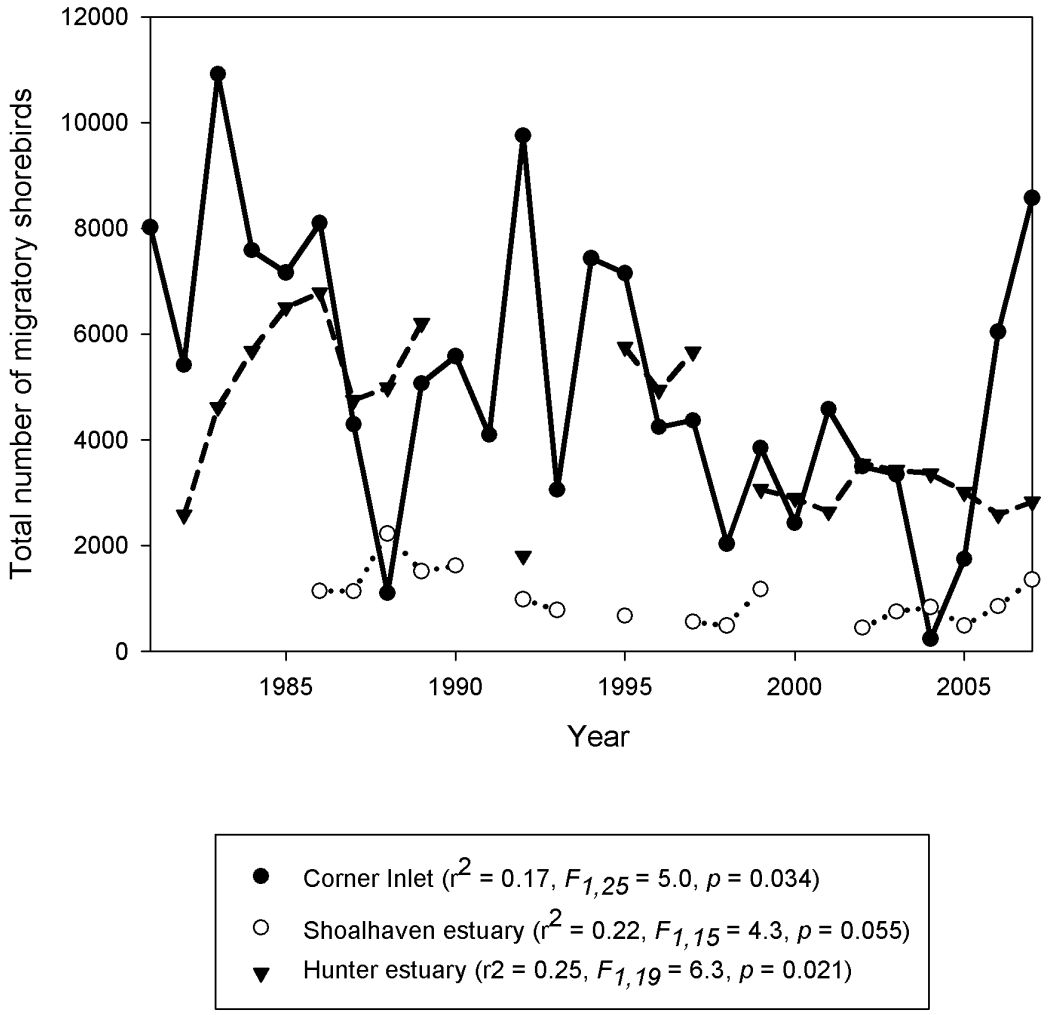


Figure 2.3 Total annual counts of migratory shorebirds in each summer (Jan/ Feb) (1981-2007) in the Corner Inlet, Shoalhaven estuary and Hunter estuary. Note that there were missing counts for the Hunter (1990, 1991, 1993, 1994 and 1998) and Shoalhaven (1991, 1994, 1996, 2000 and 2001) estuaries (see Methods).

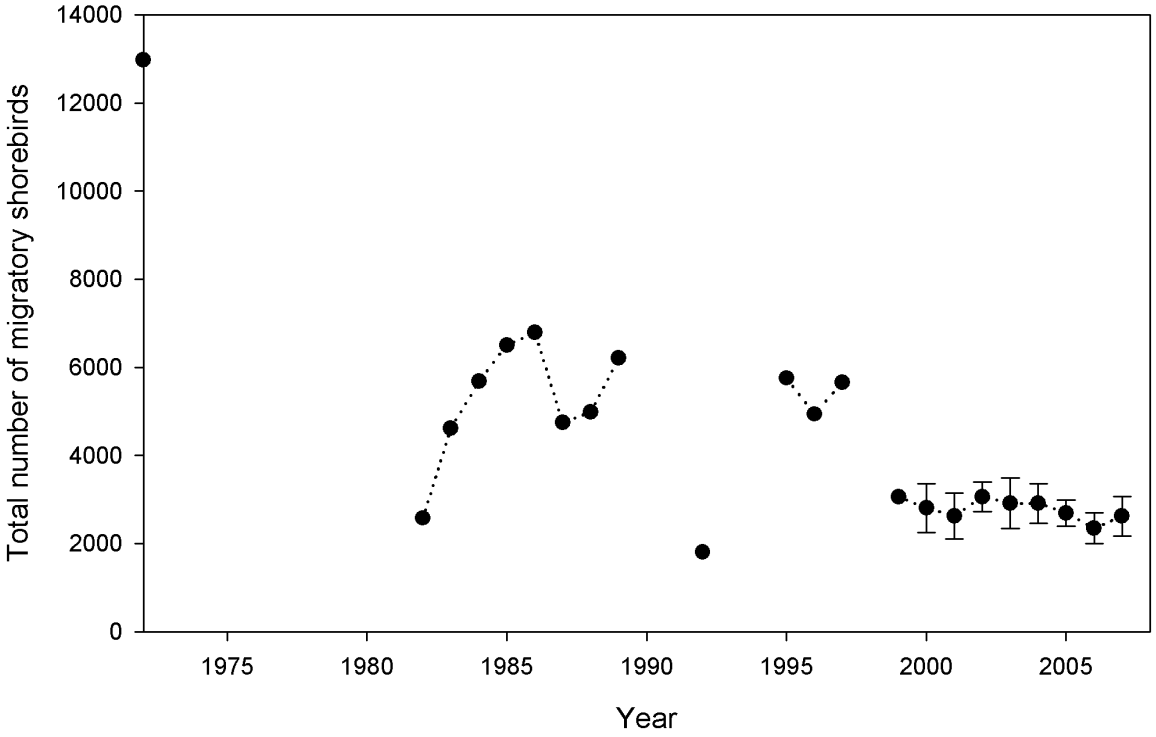


Figure 2.4 Total counts of migratory shorebirds in the Hunter estuary between 1965 and 2007. The 1970s count is an estimate of the maximum total number of migratory shorebirds present in the estuary in this period (see Table 2.3). Total counts (Jan/Feb) are presented for 1982-99 but from 2000 onwards mean counts (with 95% CI) for each year are presented for all summer months (Oct-Mar) (data supplied by AWSG/HBOC). Note that there were missing counts for 1990, 1991, 1993, 1994 and 1998 (see Methods).

Table 2.3 Mean (± standard error) and maximum counts of migratory shorebird species in the Hunter estuary in summer months (1965-2007).

Common name	1965-81	1982-85 (n = 4)		1986-90 (n = 4)		1991-95 (n = 2)		1994–97[a] (n = 8)		1996-00 (n = 4)		2004-05[a] (n = 3)		2001-07 (n = 7)	
	max	mean	max	mean	max	mean	max	mean	max	mean	max	mean	max	mean	max
Asian Dowitcher	-	0	1	0	1	0	-	0	-	0	-	0	-	0	-
Bar-tailed Godwit	3000	1305 (360)	4000	1863 (314)	2400	1486 (516)	5000	1277 (133)#	2900	1448 (135)	3100	628 (93)#	732	1067 (72)	1450
Black-tailed Godwit	800	468 (135)	800	430 (115)	618	176 (124)	400	291 (36)#	370	277 (51)	370	7 (4)#	120	190 (35)	425
Broad-billed Sandpiper	180	4 (2)	15	2 (1)	3	0	3	0	-	0	-	0	-	0.1 (0.1)	4
Buff-breasted Sandpiper	1	0	-	0	-	0	-	0	-	0	-	0	-	0	1
Common Greenshank	200	209 (119)	561	72 (26)	150	99 (41)	362	82 (14)	352	123 (13)	350	121 (29)	273	139 (24)	264
Common Sandpiper	11	0	2	1 (0.3)	2	1 (1)	3	0.3 (0.2)	2	0.3 (0.3)	2	0.3 (0.3)	2	2 (1)	6
Curlew Sandpiper	3500	1542 (255)	4000	1564 (256)	2200	985 (535)	1650	1846 (232)	2600	1325 (533)	2637	18 (10)	240	246 (34)	812
Eastern Curlew	1000	491 (70)	900	379 (62)	800	229 (171)	1000	307 (67)	750	449 (105)	600	338 (20)	673	484 (32)	786
Great Knot	40	3 (2)	7	2 (2)	8	10 (10)	20	15 (3)	50	17 (5)	50	2 (1)	7	4 (1)	60
Greater Sand Plover	31	1 (1)	3	0	-	0	1	0	-	6 (6)	23	0	-	0	2
Grey Plover	1	0	2	0	-	0	1	0	-	0	-	0	4	0	4
Grey-tailed Tattler	100	40 (20)	96	40 (5)	55	7 (5)	80	7 (5)	30	18 (5)	80	10 (3)	35	11 (4)	47
Hudsonian Godwit	-	0	1	0	1	0	-	0	-	0	-	0	-	0	-
Latham's Snipe	6	0.3 (0.3)	18	0	-	0	-	0	3	0.3 (0.3)	10	0	-	0.1 (0.1)	20
Lesser Sand Plover	500	95 (32)	200	47 (22)	84	39 (4)	47	20 (4)	170	8 (8)	35	0	16	0	6
Lesser Yellowlegs	-	0	-	0	-	0	-	0	-	0	-	0	-	0	1
Little Curlew	3	0	32	0	-	0	7	0	-	0	-	0	-	0	3
Little Stint	-	0	-	0	-	0	1	0	-	0	-	0	-	0	-
Long-toed Stint	-	0	-	0	-	0	-	0	-	0	-	0	-	0	1
Marsh Sandpiper	500	88 (55)	277	49 (26)	115	218 (216)	433	147 (35)	230	153 (17)	299	49 (9)	86	102 (17)	342
Oriental Plover	18	0	-	0	-	0	-	0	-	0	-	0	-	0	1
Pacific Golden Plover	370	413 (151)	800	445 (87)	700	147 (2)	200	96 (31)	220	79 (47)	395	115 (9)	159	158 (42)	347
Pectoral Sandpiper	25	0	10	0	5	0	1	0	-	0	10	0	-	0	5
Red Knot	600	11 (4)	1000	26 (18)	80	10 (5)	100	11 (4)	2000	10 (7)	2000	77 (29)	120	24 (11)	1669
Red-necked Stint	540	65 (26)	450	150 (10)	178	26 (26)	400	51 (23)	170	53 (29)	278	8 (8)	115	36 (12)	350
Ringed Plover	1	0	-	0	-	0	-	0	-	0	-	0	-	0	-
Ruff	5	0	1	0	-	0	1	0	-	0	1	0	4	0	1
Ruddy Turnstone	30	16 (12)	50	11 (4)	40	5 (0)	50	1 (1)	12	1 (1)	50	0	1	0	29
Sanderling	1	0	-	0	-	0	1	0	-	0	1	0	-	0	-
Sharp-tailed Sandpiper	800	36 (23)	483	505 (289)	1065	109 (109)	1200	7 (7)	300	0	400	364 (63)	995	536 (137)	1711

See following page for footnote

Table 2.3 (cont). Mean (± standard error) and maximum counts of migratory shorebirds in the Hunter estuary in summer months (1965-2007).

Common name	1965-81	1982-85 (n = 4)		1986-90 (n = 4)		1991-95 (n = 2)		1994-97[a] (n = 8)		1996-00 (n = 4)		2004-05[a] (n = 3)		2001-07 (n = 7)	
	max	mean	max	mean	max	mean	max	mean	max	mean	max	mean	max	mean	max
Terek Sandpiper	600	28 (15)	100	66 (22)	110	77 (77)	350	58 (24)	170	113 (41)	231	4 (2)	33	21 (10)	68
Wandering Tattler	1	0	1	0	-	0	-	0	-	0	1	0	-	0	-
Whimbrel	105	35 (13)	60	33 (10)	55	160 (156)	500	7 (2)	351	61 (14)	181	12 (4)	126	36 (13)	185
Wood Sandpiper	6	0	1	0	1	0	1	0	-	0	1	0	-	0	-
Total shorebirds	12975	4846 (848)	13871	5682 (488)	8671	3781 (1975)	11812	4448 (228)*	10680	4139 (687)	11105	1752 (80)	3741	3057 (148)	8600
Total species	30		28		22		26		18		24		19		28

Mean counts were determined from summer (Jan/Feb) surveys of the Hunter estuary by the AWSG/HBOC (1982-85; 1986-90; 1991-95; 1996-00; 2001-07).
Maximum counts were determined from all summer months (Sept-Apr) for 1965-2007 (see Table 2.2 for data sources).
[a]For the two additional survey periods (1994-97; 2004-05), mean counts are presented for surveys in Jan/Feb and maximum counts are presented for all other summer survey months (Sep-Apr) (# = incomplete counts; * = includes count of unidentified small migratory shorebird species) (n = number of counts)

Table 2.4 Mean (± standard error) and maximum counts of migratory shorebird species in the Hunter estuary in winter months (1965-2007).

Common name	1965-81	1982-85 (n = 4)		1986-90 (n = 5)		1991-95 (n = 3)		1994-96[a] (n = 8)		1996-00 (n = 3)		2001-07 (n = 7)	
	max	mean	max	mean	max	mean	max	mean	max	mean	max	mean	max
Asian Dowitcher	-	0	-	0	1	0	-	0	-	0	-	0	-
Bar-tailed Godwit	400	374 (38)	480	549 (29)	620	227 (29)	400	321 (39)	375	275 (63)	600	190 (27)	400
Black-tailed Godwit	80	27 (12)	53	30 (21)	110	15 (8)	30	14 (5)	43	6 (2)	9	4 (1)	30
Broad-billed Sandpiper	-	0.3 (0.3)	1	0	-	0	-	0	-	0	-	0	2
Common Greenshank	80	16 (12)	51	7 (5)	25	3 (3)	35	2 (1)	46	1 (1)	2	7 (3)	31
Common Sandpiper	1	0	-	0	-	0	-	0	-	0	-	0	-
Curlew Sandpiper	120	184 (134)	580	104 (72)	387	20 (20)	61	42 (17)	120	12 (12)	59	0.4 (0.4)	63
Double-banded Plover	260	20 (14)	60	0	-	0	-	0.4 (0.4)	3	0	50	1 (1)	180
Eastern Curlew	200	215 (59)	373	146 (22)	220	147 (48)	260	179 (27)	267	117 (61)	208	93 (11)	163
Great Knot	-	0	-	0	-	2 (2)	5	5 (1)	9	3 (3)	8	1 (1)	9
Grey Plover	1	0	-	0	-	0	-	0	-	0	-	0	-
Grey-tailed Tattler	40	14 (1)	15	2 (1)	4	0.3 (0.3)	9	0.3 (0.3)	2	2 (2)	18	2 (1)	6
Latham's Snipe	-	0	-	0	-	0	-	0	-	0.3 (0.3)	1	0	-
Lesser Sand Plover	4	0	-	0	-	0	-	0	-	0	-	0	-
Marsh Sandpiper	1	4 (2)	8	0	-	0.3 (0.3)	6	0	1	0	-	0	15
Pacific Golden Plover	5	0	-	1 (1)	4	0	-	0	-	0	-	0	2
Red Knot	-	6 (4)	8	12 (10)	50	0.3 (0.3)	1	7 (2)	15	7 (7)	21	3 (2)	14
Red-necked Stint	30	53 (46)	190	1 (1)	4	0	-	0	-	0	-	6 (4)	35
Ruddy Turnstone	-	2 (1)	5	2 (1)	7	0	8	1 (0.4)	3	0	-	0	-
Sanderling	-	0	-	0	-	0	-	0	-	0	1	0	-
Sharp-tailed Sandpiper	-	0	-	0	-	0	-	0	-	0	-	0	1
Terek Sandpiper	5	0	-	0	-	0	-	0	1	0	-	0.1 (0.1)	3
Wandering Tattler	-	0	1	0	-	0	-	0	-	1 (1)	4	0	-
Whimbrel	-	19 (6)	30	11 (7)	37	15 (15)	29	7 (4)	23	11 (3)	20	19 (8)	53
Total shorebirds	1227	933 (226)	1855	866 (179)	1469	429 (89)	844	593 (52)*	908	437 (95)	1001	329 (19)	1010
Total species	14		14		12		11		13		13		17

Mean counts were determined from winter (Jun/Jul) surveys of the Hunter estuary by the AWSG/HBOC (1982-85; 1986-90; 1991-95; 1996-00; 2001-07).
Maximum counts were determined from all winter months (May-Aug) for 1965-2007 (see Table 2.2 for data sources).
[a]For the 1994-96 monitoring project, mean counts are presented for surveys in Jun/Jul and maximum counts are presented for all other winter survey months (May-Aug).
* = includes count of unidentified small migratory shorebird species (n = number of counts).

2.4.3 Trends for selected species

Trends for individual species varied among the non-breeding sites. There were significant declines in four species in the Hunter estuary and two species in Corner Inlet and the Shoalhaven estuary (Table 2.5; Fig. 2.5). Power analyses ($\alpha = 0.05$) indicated that with the counts available, there was a high level of power (> 90%) available to detect declines of at least 80% or more in Bar-tailed Godwit, Common Greenshank and Marsh Sandpiper populations in the Hunter estuary. Due to large variation in estimates among years, the analysis was not sensitive to more subtle declines in these species. For example, the level of statistical power was too low (6 - 31%) to detect 50% declines in Bar-tailed Godwits, Common Greenshanks, Eastern Curlews, Grey-tailed Tattlers, Marsh Sandpipers or Terek Sandpipers in the Hunter estuary from 1982-2007. There were significant declines in mean counts of Curlew Sandpipers (83%), Black-tailed Godwits (56%), Lesser Sand Plovers (~99%) and Pacific Golden Plovers (67%) (Fig. 2.5; 2.6). Large flocks (>2,000 birds) of Curlew Sandpipers were observed in the Hunter estuary (1965-97), but total counts have not exceeded 500 birds since 1999 (Fig. 2.5). There were also significant declines (80%) in the total number of Curlew Sandpipers passing through the Hunter estuary in the early stages of the non-breeding season (Sept-Oct) (Table 2.3) and in Curlew Sandpipers recorded further south in Corner Inlet (88%) (Table 2.5). The Hunter estuary's Black-tailed Godwit population also declined significantly from 449 (± 82 S.E.) birds in the 1980s to 196 (31 ± S.E.) birds in 2001-07 (Fig. 2.5; Table 2.3). Maximum counts of Lesser Sand Plover in the Hunter estuary now represent 3% of counts reported during 1965-81 (Table 2.3). Lesser Sand Plovers were recorded regularly in annual summer surveys of the Hunter estuary but maximum counts were low (16 birds) between 2000-07. Maximum counts of Pacific Golden Plover ranged between 700 - 800 birds in the 1980s but have not exceeded 400 since 1990 (Table 2.3). Common Greenshanks declined in Corner Inlet and the Shoalhaven estuary and Eastern Curlew counts declined by over 60% in the Shoalhaven estuary (1986 - 2007) (Table 2.5).

Table 2.5 Mean annual summer counts (± standard error) of six migratory shorebird species at the non-breeding sites (1981-2007), showing correlations with time, *F* ratios and probabilities of trends.

Site		n	Bar-tailed Godwit	Curlew Sandpiper	Common Greenshank	Eastern Curlew	Lesser Sand Plover	Pacific Golden Plover
Corner Inlet	1981-89	9	836 (227)	1822 (550)	68 (16)	279 (60)	0 (0)	0 (0)
	1990-99	10	725 (155)	1519 (539)	124 (25)	387 (87)	0.2 (0.2)	2 (2)
	2000-07	8	632 (157)	221 (63)	32 (8)	350 (107)	0 (0)	0 (0)
	r^2		0.02	0.21	0.19	0.00	-	-
	F		0.6	6.5	6.0	0.0	-	-
	p		0.440	**0.017**	**0.022**	0.964	-	-
Hunter estuary	1982-89	8	1584 (245)	1553 (167)	141 (62)	435 (48)	71 (20)	429 (81)
	1990-99	5	1484 (192)	1388 (400)	116 (18)	335 (102)	21 (9)	119 (35)
	2000-07	8	1101 (71)	257 (31)	136 (21)	496 (30)	0 (-)	140 (40)
	r^2		0.04	0.78	0.08	0.01	0.51	0.26
	F		0.8	29.9	1.5	0.3	19.5	6.7
	p		0.382	**<0.001**	0.230	0.607	**<0.001**	**0.018**
Shoalhaven estuary	1986-89	4	646 (225)	2 (0.4)	14 (6)	183 (28)	10 (5)	210 (12)
	1990-99	7	482 (95)	2 (2.1)	4 (2)	73 (28)	9 (7)	98 (29)
	2000-07	6	353 (37)	0.2 (0.2)	0.3 (0.3)	67 (21)	13 (8)	191 (33)
	r^2		0.10	-	0.34	0.27	0.02	0.00
	F		1.6	-	7.8	5.4	0.3	0.0
	p		0.226	-	**0.014**	**0.034**	0.610	0.944

The results of linear regression analyses of the transformed data are presented. This analysis was based on annual Jan/Feb counts only (n = number of counts). Data supplied by AWSG/HBOC and NSW NPWS (see Methods).

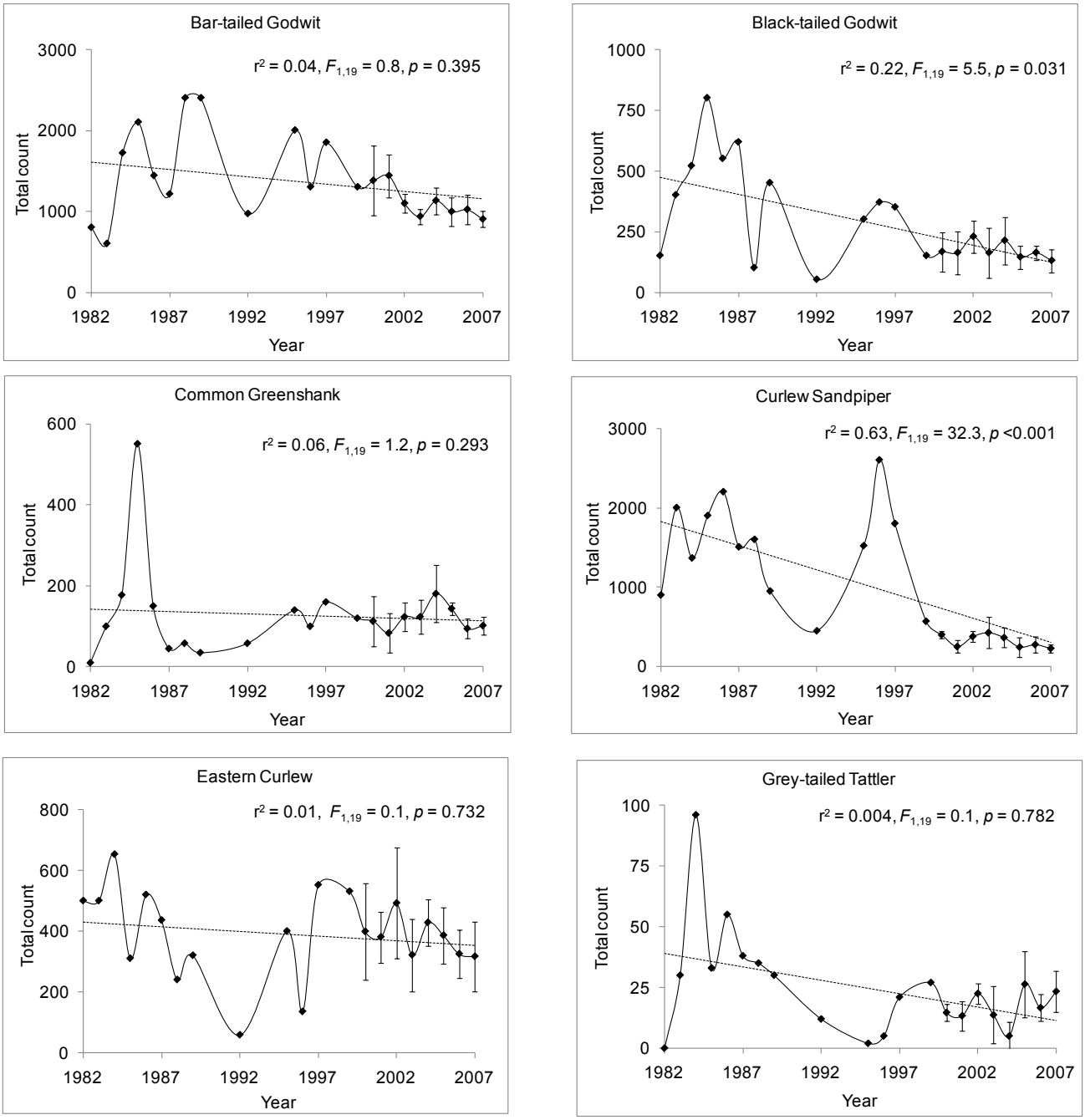


Figure 2.5 Mean counts (± 95% CI) of Bar-tailed Godwit, Black-tailed Godwit, Common Greenshank, Curlew Sandpiper, Eastern Curlew and Grey-tailed Tattler in the Hunter estuary in summer surveys from 1982-2007 (data provided by AWSG/ HBOC). Multiple summer counts were available from 1999. Results of linear regression analyses of the transformed counts are presented (dotted lines show direction of trends).

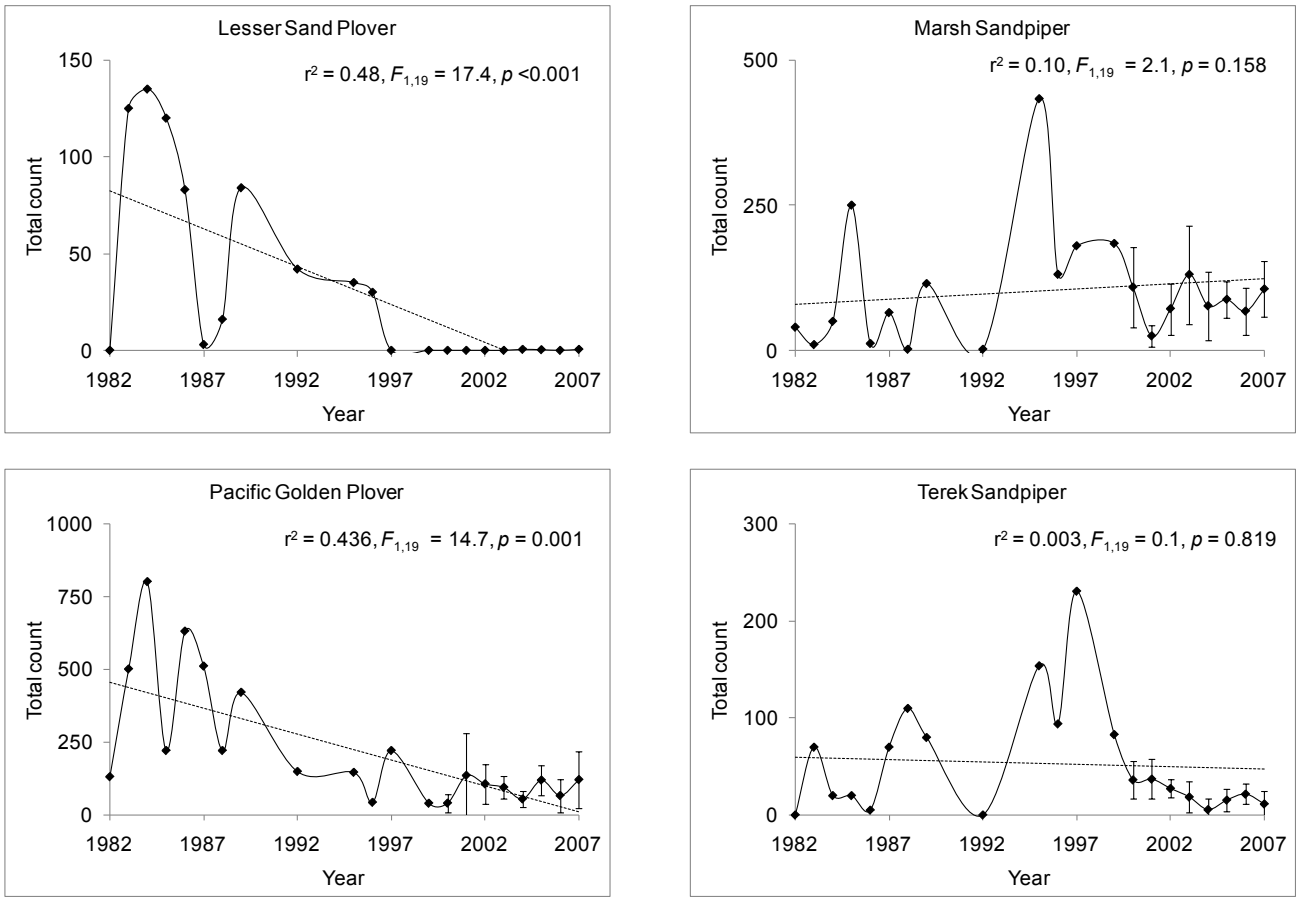


Figure 2.6 Mean counts (± 95% CI) of Lesser Sand Plover, Marsh Sandpiper, Pacific Golden Plover and Terek Sandpiper in the Hunter estuary in summer surveys from 1982-2007 (data provided by AWSG/ HBOC). Multiple summer counts were available from 1999. Results of linear regression analyses of the transformed counts are presented (dotted lines show direction of trends).

2.5 Discussion

2.5.1 Population trends

There is increasing evidence that shorebird populations are declining worldwide (Zöckler *et al.* 2003; Wetlands-International 2006). Of the 41% of shorebird populations with known trends (210 populations) in the world, 48% are in decline and only 16% are increasing overall. Australia is in the East Asian-Australasian Flyway (EAAF), which has unknown trends for 85% of its shorebird populations and of the 12 populations in the flyway with known trends, nine may be in decline (Stroud *et al.* 2006). In over 25 years, total numbers of migratory shorebirds have declined significantly at Corner Inlet (41%), and the Shoalhaven (48%) and Hunter (42%) estuaries (1981-2007) (Fig. 2.3). Despite missing count data for several summers in the 1990s, there was still evidence for decline in migratory shorebird numbers from 1990 to 2007 (Fig. 2.4). Previous estimates of the total number of shorebirds visiting the estuary in the 1970s ranged widely from 6,450 - 8,620 (Gosper 1981), 7,900 (Kingsford & Ferster Levy 1997), 8,000 - 10,000 (van Gessel & Kendall 1974) to 15,000 - 20,000 birds (van Gessel 1976) but in the 1980s, the total number of migratory shorebirds had declined to 5,219 - 5,300 birds (Morris 1983; Kingsford & Ferster Levy 1997). This trend is consistent with my analysis which included an additional 17 years of count data. Total numbers of shorebirds declined from around 5,264 birds in the 1980s and 4,244 birds in the 1990s to 3,036 birds in 2000-07. If this rate of decline (1.6% yr^{-1}) is sustained, there would fewer than 1,000 migratory shorebirds in the Hunter estuary by 2050.

Based on estimates of population sizes in 1970-90, the Hunter estuary previously supported 11 migratory shorebird species in numbers > 1% of their Australian populations and seven species in internationally significant numbers (> 1% flyway population) (Smith 1991; Watkins 1993) but this is no longer the case (Table 2.6). At a national scale, the Hunter estuary now only supports five of the original 11 species listed in Watkins (1993) in numbers > 1% of Australian populations: the Common Greenshank, Eastern Curlew, Marsh Sandpiper, Pacific Golden Plover and Whimbrel *Numenius phaeopus*. Based on recent maximum counts (2001-07), the Hunter estuary supports two species in internationally significant numbers; the Eastern Curlew (2% of flyway population) and Sharp-tailed Sandpiper (1% of flyway population). Prior to 1990, Sharp-tailed Sandpipers had not been recorded in internationally or nationally significant numbers in the Hunter estuary. In the period 2001-07, maximum counts of Red Knots (1,669 birds) in the Hunter estuary also exceeded 1% of the estimated Australian population. The Hunter estuary, however, no longer supports more

than 1% of flyway populations of six species: the Broad-billed Sandpiper, Bar-tailed Godwit, Common Greenshank, Curlew Sandpiper, Lesser Sand Plover or Terek Sandpiper, which were previously listed in a review by Watkins (1993) (Table 2.6).

Table 2.6 Maximum counts of species in the Hunter estuary which have occurred in excess of 1% of their Australian or flyway populations (1970 - 2007).

Common name	Hunter estuary		Australia[c]	Flyway[d]	Status[e]
	1970-90[a]	2001-07[b]			
Bar-tailed Godwit	4 000	1450	165 000	325 000	- (I)
Black-tailed Godwit	800	425	81 000	160 000	- (N)
Broad-billed Sandpiper	180	4	8 000	25 000	- (I)
Common Greenshank	561	273	20 000	60 000	N (I)
Curlew Sandpiper	4 000	812	155 000	180 000	- (I)
Eastern Curlew	653	786	29 000	38 000	I (I)
Lesser Sand Plover	800	16	24 000	140 000	- (I)
Marsh Sandpiper	500	342	9 000	100 000 - 1 000 000	N (N)
Pacific Golden Plover	800	347	7 300	100 000 - 1 000 000	N (N)
Red Knot	-	1 669	135 000	220 000	N (-)
Sharp-tailed Sandpiper	-	1 711	140 000	160 000	I (-)
Terek Sandpiper	600	68	25 000	60 000	- (I)
Whimbrel	105	185	10 000	100 000	N (N)

[a] Maximum counts of species originally recorded in significant numbers in the Hunter estuary 1970-90 (from Watkins 1993)
[b] Maximum counts in the Hunter estuary from 2001-07 (based on Table 2.3).
[c] Current population estimates for Australia were based on DEH (2005)
[d] Population estimates for the EAAF were taken from Bamford *et al.* (2008).
[e] Status: I = International importance (1% of flyway population), N = National importance (1% of Australian population) (NB all internationally important sites are also nationally important). Original listings from Watkins (1993) are presented in parentheses.

There has been a decline in the reporting rate of at least seven migratory shorebird species across south-eastern Australia, between the first (1977-81) and second (1998-02) national Birds Australia atlases (Barrett *et al.* 2003), with declines of up to 49% reported for some species. The species most affected have been the Black-tailed Godwit, Curlew Sandpiper, Double-banded Plover *Charadrius bicinctus*, Great Knot *Calidris tenuirostris*, Pacific Golden Plover, Red-necked Stint and Terek Sandpiper (Olsen & Weston 2004). It may be that declines in species more common in northern Australia are more visible in southern Australia, as species may contract into their preferred range following population declines. The Black-tailed Godwit, Broad-billed Sandpiper, Greater Sand Plover, Grey-tailed Tattler, Lesser Sand Plover and Terek Sandpiper, for example, are most common in northern Australia while fewer birds are generally found in NSW (Lane 1987; Smith 1991; Watkins 1993). Maximum counts of all six species were larger in the Hunter estuary in the 1970s than in the following decades (Table 2.3) and similar declines in these species have

been observed in other parts of south-eastern Australia (Olsen & Weston 2004; Gosbell & Clemens 2006).

Much of the recent decline (from 1999) in mean total numbers of migratory shorebirds in the Hunter estuary was due to significant declines (83%) in Curlew Sandpipers. This species has experienced widespread declines across south-eastern Australia, including Corner Inlet (88%) (Table 2.5), Coorong in South Australia (89%) (Wilson 2001) and other non-breeding sites in Victoria, south-east Tasmania, South Australia and south-west Western Australia (Barter 1992; Creed & Bailey 1998; Wilson 2001; Olsen *et al.* 2003; Gosbell & Clemens 2006). In a review of the national shorebird monitoring program, Curlew Sandpipers declined in all 11 of the sites analysed, with significant average declines of 3% to 4% per year observed at nine monitoring sites (Gosbell & Clemens 2006). Historically, Curlew Sandpipers were one of the most common species in the Hunter estuary, with summer counts regularly exceeding 1,500 birds and smaller flocks sometimes remaining over winter months (37-580 birds). Although relatively low numbers were recorded in annual summer surveys in 1982 (900 birds), 1989 (950 birds) and 1992 (450 birds), numbers of Curlew Sandpipers recovered in subsequent years (Fig. 2.5). However, since 1999, annual summer counts of Curlew Sandpipers have not exceeded 570 birds and this species has also been absent from annual winter counts.

Declines in some species have been site specific. In this study, for example, Eastern Curlew populations did not change significantly in the Hunter estuary and Corner Inlet, but this species declined by over 60% in the Shoalhaven estuary and similar declines have been seen in other sites in South Australia and south-east Tasmania (Close & Newman 1984; Thomas 1987; Reid & Park 2003). Although there were no significant trends in Bar-tailed Godwit populations in the Hunter estuary, Corner Inlet or Shoalhaven estuary in this study, national trends suggest that this species may be declining across southern Australia (Gosbell & Clemens 2006). Anecdotal reports suggest that declines in Lesser Sand Plover were not limited to the Hunter estuary, as similar declines have also been recorded in Botany Bay, Sydney (Morris & Burton 1995) and in the Shoalhaven estuary, which supported large numbers of Lesser Sand Plovers in the 1970s (NPWS 1998b). Declines in the Lesser Sand Plover, Broad-billed Sandpiper and Greater Sand Plover may have occurred in the Hunter estuary before regular counts began in the 1980s in response to the removal of sandflats and islands by major dredging of the river channel in preceding years.

2.5.2 Difficulties with detecting trends

The detection of real trends in numbers of migratory shorebird species is affected by errors introduced in counting and by using multiple data sets collected by different observers (see Appendix A1). Shorebird counts tend to underestimate true population sizes but the relative error is probably low (< 10%) (Hale 1974; Underhill & Prys-Jones 1994). The most serious errors result from missing birds, either through a visibility bias or lack of precision (Rappoldt *et al.* 1985). This is difficult to quantify, particularly when the locations of some high tide roosts are unknown. Counts of Double-banded Plovers, Pacific Golden Plovers and Ruddy Turnstones in the Hunter estuary may have been affected by changes in their preferred high tide roost sites. Large flocks of Double-banded Plovers were observed in the estuary in the 1970s (165-260 birds) (van Gessel & Kendall 1974; Stuart *in prep.*) and 180 birds were seen on intertidal mudflats in Fullerton Cove in June 2004 (Stuart 2005). As this species generally shows a high degree of site fidelity in its non-breeding range (Barter & Minton 1987), such isolated records of large flocks suggest that either roosting locations exist outside the main survey area or that Double-banded Plovers only use the Hunter estuary as a stop-over site during their migration. Ruddy Turnstones have only been observed in small numbers (1-3 birds) in monthly surveys since 1999 (Table 2.3), but the evidence for decline in this species is also inconclusive as larger flocks (up 30 birds) can roost on rock platforms near the entrance of Newcastle harbour (Fig. 2.2), which are not counted in regular surveys of the estuary. Pacific Golden Plovers may also have been underestimated in regular surveys (158 ± 42 SE) (2001-07), as large flocks (>300 birds) have been seen foraging in the estuary in recent years (Crawford & Herbert 2007; J. Spencer pers. obs.) (Table 2.3).

Single counts for a given year are problematic as additional counts provide more confidence under normal counting conditions (Underhill & Prys-Jones 1994), identifying potentially erroneous counts. A count of 180 Broad-billed Sandpipers in February 1972 (van Gessel & Kendall 1972a) was exceptionally high, given that no counts of more than 15 birds were observed from 1982-2007 (Table 2.3). Although maximum numbers of migratory shorebirds were similar across summer months from 1965 - 2007, these estimates were inflated by observations of large flocks of Bar-tailed Godwits, Curlew Sandpipers, Red Knots and Red-necked Stints (Table 2.3), which migrate through the estuary during the early stages of their non-breeding season (Sept-Oct). Large flocks of Terek Sandpipers (400-600 birds) were also observed on at least three occasions in the early 1970s (Holmes 1970; van Gessel & Kendall 1972a; Rogers 1973) and flocks of 350 and 231 Terek Sandpipers were observed in annual summer counts in the estuary in 1992 and 1997, respectively

(Morris & Burton 1994; Harris 1997). It is unclear, however, whether these observations were of migrating flocks or of birds that remained in the estuary throughout the non-breeding season.

For the ten species analysed in most detail, high levels of variability among counts of some species resulted in low statistical power to detect population change. In the Hunter estuary, high variability in counts of Bar-tailed Godwits, Common Greenshanks and Marsh Sandpipers meant that only very large (> 80%) population declines could be detected with the counts available from 1982-2007 and population declines greater than 90% had to occur before significant trends were detected for Eastern Curlews, Grey-tailed Tattlers and Terek Sandpipers.

2.5.3 Causes of decline

Many factors may have contributed to declines in shorebird populations in the Hunter estuary and other non-breeding sites in southern Australia. It is difficult to determine whether declines are a result of local changes at non-breeding sites, which may make these sites less attractive to shorebirds, or if declines are an indication of habitat loss in other parts of the flyway or simply poor breeding success in previous seasons. Fluctuations in counts of some migratory shorebirds can reflect the success of previous breeding seasons (Minton *et al.* 2003), which can vary in response to weather conditions and levels of predation at breeding sites in the Arctic (Underhill *et al.* 1993; van de Kam *et al.* 2004). There was evidence for declines in the number of juvenile birds (61% decline in mean total numbers) remaining in the Hunter estuary over winter months and this could reflect negative factors operating on the breeding or migratory staging grounds.

The scale of decline in Curlew Sandpiper populations in southern Australia, however, indicates that habitat loss and degradation at key stop-over sites is a more likely cause of population declines for this species, as recent evidence suggests that declines in Curlew Sandpiper populations may be a result of reduced adult survival rates rather than recruitment failures (Rogers & Gosbell 2006). The Yellow Sea is a critical staging site for many shorebird species, including the Curlew Sandpiper (Minton *et al.* 2006), which spends its non-breeding season in Australia. However, large-scale reclamation projects along the Chinese and Korean coastlines have destroyed huge areas of intertidal mudflats, which has displaced many shorebirds from key staging sites in the Yellow Sea (Barter 2002; Moores 2006; Moores *et al.* 2007).

More than 200 years of industrial and urban development have caused massive changes to the Hunter estuary and although no counts were available prior to 1965, significant declines in shorebird habitat and populations probably occurred before records began. It is highly probable that the original islands, and surrounding mudflats and sandbanks, which were removed by dredging of the main river channel, would have supported many more shorebirds during their non-breeding season than seen in the estuary today. The foreshore of many non-breeding sites in southern Australia has been developed for residential and industrial use, and remaining habitats can experience high levels of disturbance from recreational activities (Taylor & Bester 1999; Blumstein *et al.* 2003; Foate 2005). Although the availability of high tide roosting habitat has been identified as a limiting factor for shorebirds in the Hunter estuary (Geering 1995; Kingsford & Ferster Levy 1997; Howe 2008), the loss of supplementary high tide feeding habitat in the estuary may also have contributed to local declines in Curlew Sandpipers and other small shorebirds. Small shorebird species are often most affected by habitat loss as they sometimes need to feed continuously through the tidal cycle, and often make extensive use of higher flats (Goss-Custard & Moser 1988) or artificial wetlands during high tide periods (Masero *et al.* 2000; Masero & Perez-Hurtado 2001; Masero 2003). The Stockton sewage treatment works, which ceased operations in 2002, and 'Big Pond' (Fig. 2.2), which was drained in the late 1990s, formerly supported large numbers of small shorebird species (Geering 1995; Straw 1999). For example, flocks of 1,500 Curlew Sandpipers were observed at the sewage works during spring high tides in 1992-93 (Geering & Winning 1993).

More insidious changes upstream of the Hunter estuary may also have degraded foraging habitats. The loss of flows to important wetlands in southern Australia is thought to have caused major declines in migratory and Australian breeding resident shorebird species in eastern Australia from 1983-2006 (Nebel *et al.* 2008). Declines in Curlew Sandpiper numbers in the Coorong, for example, have occurred alongside reductions in flows and increased salinity at the mouth of the Murray River (Gosbell & Grear 2005; Nebel *et al.* 2008). It is unclear whether the extraction of freshwater for irrigation and dams upstream of the Hunter and Shoalhaven estuaries has had similar impacts on the ecological health of intertidal habitats and shorebird populations. Plans for a new dam in the upper Hunter valley (HWC 2007) will reduce flows and sediment inputs in the Hunter estuary further, which may put additional pressure on remaining foraging habitats.

2.5.4 Conclusions

The cumulative effect of wetland loss and degradation in Australia and other parts of the flyway is clearly the main threat to migratory shorebird populations. Shorebirds can be extremely site faithful (Dann 1981a; Driscoll 1995; Rehfisch *et al.* 1996), therefore, habitat loss can directly impact their survival and fitness (Burton *et al.* 2006). There is strong evidence that shorebird populations and their habitats are in decline in the Hunter estuary and other parts of the flyway. The challenge is now to conserve and manage remaining shorebird habitats. In recognition of the Hunter estuary's importance at both a national and flyway scale, rehabilitation efforts have focused on creating and rehabilitating shorebird habitat. These efforts, however, may prove futile, given the resurgence of development proposed for the Hunter River, which includes a large dam in the upper catchment, and further dredging and widening of the main channels, extension of coal loading facilities and the construction of a major transport corridor in the lower estuary. A more sustainable alternative would be for Australia to embrace its responsibility to protect nationally significant wetlands for shorebirds and to lead international efforts to halt the destruction of staging sites in other parts of the flyway.

CHAPTER 3: ROOSTING BEHAVIOUR AND DISTURBANCE RATES OF THE EASTERN CURLEW *NUMENIUS MADAGASCARIENSIS* AT HIGH TIDE ROOSTS

3.1 Abstract

Shorebirds rest at roost sites at high tide when their low tide feeding habitat is unavailable. Coastal development and excessive disturbance in some estuaries has resulted in the loss of shorebird roosts and declines in shorebird populations. In this chapter, I investigated roosting behaviour and disturbance rates of a migratory shorebird, the Eastern Curlew *Numenius madagascariensis* during one non-breeding season in the Hunter estuary, south-eastern Australia (2005-06). Observations of shorebirds were made at two day roosts, an artificial rock wall and sandspit, while an area of flooded saltmarsh was observed during night-time high tides. Eastern Curlews spent over 50% of their time in a resting posture at the roost sites, while vigilance behaviour represented about 30 - 40% of their roosting time. At the day roosts, vigilance levels were raised (by 18 - 20%) prior to their migration (Feb-Mar). Resting time in focal birds increased with time spent at the rock wall but did not change over the tidal period at the other roosts. Microhabitat use was site specific. Shallow water was important at the sandspit and night roost. Birds were disturbed frequently at the day roosts (0.8 - 1.7 hr^{-1}), but a larger proportion of the flock took flight following a disturbance at the sandspit (82%) compared to the rock wall (46%). Birds of prey were the most common cause for birds to take flight but birds spent longer in flight after being disturbed by fishers and walkers. The viability of shorebird roosting habitat in the Hunter estuary continues to be threatened by port development, human-induced disturbance, erosion, mangrove encroachment and sea level rise.

3.2 Introduction

In coastal wetlands, the loss and degradation of roosting habitat can directly impact on shorebird populations, as roosting takes up to 50% of their daily activity (Burton *et al.* 1996). Shorebird population declines have been linked to development and excessive disturbance of roost sites in the United Kingdom (Mitchell *et al.* 1988; Burton *et al.* 1996) and United States (Pfister *et al.* 1992). In south-eastern Australia, shorebird habitats can also be heavily disturbed where foreshores have been developed for industrial and residential use (Kingsford & Ferster Levy 1997; Harding *et al.* 1999; Taylor & Bester 1999; Blumstein *et al.* 2003).

Shorebirds use roost sites at high tide to rest, preen and bathe while their low tide feeding habitat is inundated. Roost sites are usually above the mean high water mark and can include rock walls, sandspits, oyster leases, saltmarsh and ocean beaches (Lane 1987). Choice of roost site is determined by predation risk, levels of disturbance and energetic costs at each roost site (Lawler 1996; Luis *et al.* 2001; Rogers *et al.* 2006a) (Table 3.1). Although the relative importance of each component is not clear, the fitness of birds is assumed to indicate the quality of a given roost site (Luis *et al.* 2001).

A key factor in roost choice is predation risk (Table 3.1). When predation risk is high, some shorebirds flock over the ocean during the high tide period rather than settle at available roost sites (Dekker 1998; Hotker 2000). Shorebirds generally use different roost sites during day and night with changing predation risks (Swineboard 1961; Handel & Gill 1992; Rohweder 2001; Rogers *et al.* 2006a). During day-time high tides, most shorebirds roost close to their low tide intertidal feeding habitat (Lawler 1996; Dias *et al.* 2006), while at night they often roost in remote locations that are less vulnerable to ground predators such as foxes and cats (Handel & Gill 1992; Sitters *et al.* 2001; Rogers *et al.* 2006b). Day-time roosts generally have an open aspect, which allows shorebirds to detect birds of prey (Page & Whitacre 1975; Cresswell 1994; Dekker & Ydenberg 2004). In one study, Peregrines *Falco peregrinus* were most successful when they could use vegetation as a screen to ambush roosting shorebirds (Dekker 1998).

Fishers, walkers, dogs, watercraft and machinery can also disturb shorebirds at their day roosts (Table 3.1). High levels of disturbance by human activity and avian predators can affect the survival and fitness of shorebirds (Durell *et al.* 2005; Goss-Custard *et al.* 2006), however, their

tolerance to disturbance varies among species (Furness 1973; Blumstein *et al.* 2003; Durell *et al.* 2005), times of day (Burger & Gochfeld 1991) and roost sites (Rogers *et al.* 2006b). The frequency of disturbance and distance at which shorebirds take flight are often the quantified measures of disturbance (Burger 1981; Taylor & Bester 1999; Blumstein *et al.* 2003). A more subtle measure of disturbance is the level of vigilance and sleep behaviour in roosting individuals. Many bird species sleep with one eye open, so they can respond quickly to perceived threats. Sleep is often accompanied by periods of eye closure interrupted by short periods of eye opening or 'peeking' (Lendrem 1983, 1984; Rattenborg *et al.* 1999). Few studies, however, have investigated the sleep-vigilance trade-off in shorebirds at high tide roosts (Dominguez 2003).

Physical characteristics (e.g., topography, elevation and substrate) are also important for determining the suitability of roosting habitat (Table 3.1). For example, shorebirds prefer wet substrates or shallow water to keep cool during high ambient temperatures (Rogers *et al.* 2006b; Amat & Masero 2007). Proximity to water is also important for some shorebirds which depend on high tide feeding because of insufficient food intake during the previous low tide feeding period (Luis *et al.* 2002).

Table 3.1 High tide roost characteristics and their importance to shorebirds (Sources: Lawler 1996; Luis *et al.* 2001; Rogers 2003).

Component	Roost characteristic	Importance to shorebirds
Energetic costs	Topography/elevation	Protection from adverse weather
	Substrate texture and hardness	For cooling/availability of foraging habitat
	Proximity to feeding areas	Travelling time to and from foraging habitat
Predation risk (vigilance)/ energetic costs	Vegetation cover/type	Protection from adverse weather/ camouflage from predators
	Availability of shallow water	For cooling/availability of foraging habitat and protection from ground predators
	Distance to tall vegetation (visibility)	Tall vegetation provides cover for predators
	Proximity to foreshore	Escape distance from predators
	Roost background colour	Conspicuousness to predators
	Remoteness	Background noise from vehicles or machinery/access for ground predators
Disturbance/ energetic costs	Size of roost	Levels of inter or intra-specific aggression
	Distance to alternative roosts	Time spent in flight
	Presence of people/predators	Time spent in flight

In this chapter, I investigated the roosting behaviour, microhabitat use and disturbance rates of Eastern Curlews *Numenius madagascariensis* in 2005-06, at two major day roosts and one night roost in the Hunter estuary, south-eastern Australia. The availability of roosting habitat is thought to be limiting shorebird populations in this non-breeding site (Geering 1995; Kingsford & Ferster Levy 1997), as the Hunter estuary has undergone significant changes since its development as a major port in the early 1900s (see Chapter 2 for a review). Night roosting habitat has been most affected, with the drainage of saltmarsh for agricultural land (Clarke & van Gessel 1983) and encroachment of mangrove into remaining areas of saltmarsh (Williams *et al.* 2000). I focused on the Eastern Curlew as it is relatively common in the Hunter estuary, with about 400 birds recorded regularly during austral summer (Oct-Mar). Maximum counts (786 birds) of Eastern Curlews recorded in the Hunter estuary represent more than 2% of this species' population in the East Asian-Australasian Flyway (see Chapter 2).

3.3 Methods

3.3.1 Study site

This study was done in the Hunter estuary, New South Wales (NSW), Australia (32^{o} 51'S/ 151^{o} 46'E), a migratory shorebird non-breeding site adjacent to a major coal export terminal and the city of Newcastle (Fig. 3.1). A large portion of the estuary (4,255 ha) is reserved under the Hunter Wetlands National Park (DECC 2007) and part of this site (2,206 ha) was listed under the Ramsar Convention in 1984 and as a site in the East Asian-Australasian Shorebird Site Network in 1996. Tidal cycles are semidiurnal with usually one high tide during day and night periods. Surveys were done during high tides that ranged from 1.07 m (neap) to 2.01 m (spring) tides (BOM 2005).

Most migratory shorebird species in the Hunter estuary roost on the Kooragang dykes (Fig. 3.1) during day-time high tides (Geering 1995; Kingsford & Ferster Levy 1997; Straw 1999). This river training wall is about 1.6 km in length and was created in the late 1960s following major dredging works in the main river channel (Geering 1995). Over time, large sections of the rock wall have eroded, which have made it difficult for ground predators and people to access the rock wall over the high tide period and made this site more attractive as roosting habitat for shorebirds (Fig. 3.2). Stockton sandspit (Fig. 3.1) is also a major day roost in the estuary covering about 5.3 ha (Howe 2008) and was formed during the construction of Stockton bridge over the Hunter River in 1971. The Kooragang Wetland Rehabilitation Project (KWRP) and NSW National Parks and Wildlife Service (NPWS) manage the sandspit as shorebird roosting habitat. Mangroves *Avicennia marina* and invasive bitou bush *Chrysanthemoides monilifera* and spiny rush *Juncus acutus* were removed from this site to restore roosting habitat for shorebirds in 2002. The sandspit contains an artificial lagoon, fringed by sandflats and low-level saltmarsh (*Sarcocornia quinqueflora, Sporobolus virginicus)*, and is connected to the Hunter River via a rock weir (Fig. 3.2). During night-time high tides, many shorebirds move to Windeyers Reach on Kooragang Island on the North Arm of the Hunter River (Fig.3.1) (Geering 1995; Foate 2005). This roost site covers 7.5 ha (Howe 2008) and contains two shallow ponds, flooded during spring high tides and is fringed by saltmarsh and tall (> 2 m) mangroves (Fig. 3.2).

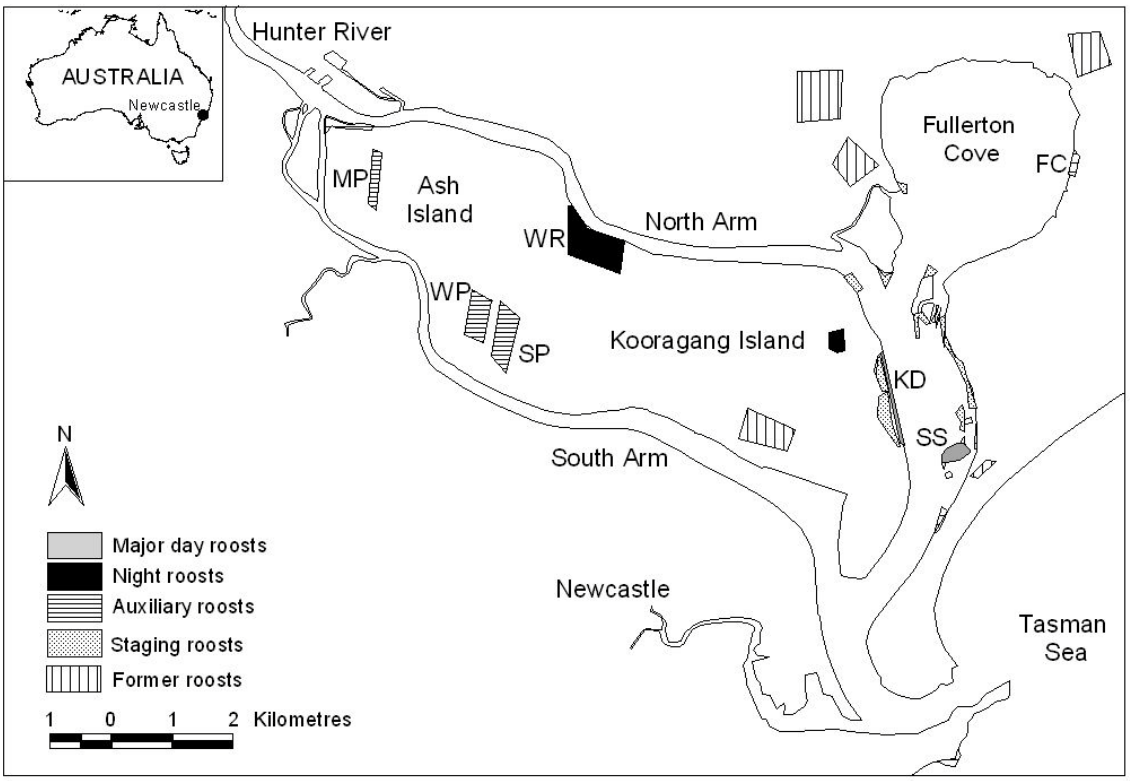


Figure 3.1 Location of the Hunter estuary, south-eastern Australia and high tide roost sites. The major day roosts, the Kooragang dykes (KD) rock wall and Stockton sandspit (SS), are in the main river channel. The main night roost, Windeyers Reach (WR), is located on Kooragang island beside the North Arm of the Hunter River. Staging and auxiliary roosts are also indicated. Saltmarsh ponds (Milhams Pond and Phoenix Flats (MP), Wader Pond (WP) and Swan Pond (SP)) on Ash Island provide supplementary day and night roosting habitat for Eastern Curlews. Eastern Curlews also roost occasionally on the eastern shore of Fullerton Cove (FC) during day time high tides.

3.3.2 Flock scans

I surveyed Eastern Curlews at the roost sites from October – early December 2005 and from February – March 2006. Eastern Curlews were observed at the main day roosts in both survey periods but night observations were restricted to the latter period (Table 3.2). I randomly assigned sampling dates for each roost site and stratified sampling of the day roosts into weekends (including public holidays) (five surveys for each site) and weekdays (11 surveys for each site) (Oct-Dec 2005; Feb-Mar 2006). I used high tide counts collected by the Hunter Bird Observers Club (HBOC) (see section 2.3.2; Appendix C) during monthly surveys from October 2005 – March 2006 (n = 6), to determine the relative importance of the major day roosts for Eastern Curlews.

Credit: C. Herbert (upper) and S. Rostas (centre and lower).

Figure 3.2 Major roost sites in the Hunter estuary: ground view of Kooragang dykes rock wall at low tide (upper); aerial view of Stockton sandspit (centre); and aerial view of the night roost at Windeyers Reach with Newcastle in the distance (lower). Observations at the night roost were restricted to the pond nearest the North Arm of the Hunter River (indicated by circle).

Table 3.2 Summary of roost surveys (Oct 2005 - Mar 2006), using instantaneous flock scans and focal sampling.

Type	Site	Season	Survey days	Behaviour Scans	Microhabitat scans	Focal samples
Day	Kooragang dykes	Oct-Dec 2005	9	134	134	144
		Feb-Mar 2006	7	74	74	109
Day	Stockton sandspit	Oct-Dec 2005	7	114	117	163
		Feb-Mar 2006	9	103	104	202
Night	Windeyers Reach	Feb-Mar 2006	8	50	51	86

I observed birds on the rock wall (Kooragang dykes) using binoculars (Swarovski 8 x 30 mm) from a four metre stationary inflatable boat moored about 60 - 100 m away in the main river channel. Observations of the rock wall were based on large flocks observed in the middle section of the rock wall (about 760 m in length or 50% of the roost), where Eastern Curlews concentrated. Birds habituated to the presence of the boat, as small recreational fishing boats regularly passed at this distance from the rock wall and birds generally only flushed when boats moved inside a navigational marker, less than 40 m from the rock wall. Boat-based observations were restricted to good weather conditions (wind speeds < 15 knots; sea states < 2 Beaufort scale). At the sandspit, I observed Eastern Curlews using a telescope (Swarovski 20 – 60X magnification) and binoculars from a bank about 100 - 200 m from roosting birds. Observations at the sandspit included all birds surrounding the main lagoon but excluded sandflats in front of the main roost (Fig. 3.2), which were not visible. These sandflats were not used by Eastern Curlews except in late stages of the roosting period before departing to their feeding sites. Observations of day roosts were about two hours before and about four hours after published high tides. Adverse weather prevented late stages of some surveys of the rock wall from being completed.

At the night roost, I made nocturnal observations from a hide (2.5 x 2.5 m dimensions) installed a month before surveys, so that birds could be habituated. Birds roosted 5 - 100 m from the hide during night time high tides. I used a night vision viewer (ITT Night Quest Series F5000, 3.0 x and 2.0 x high resolution telephoto lenses Roanoke: Virginia, USA) with a spot light and infra-red screen (Night Vision Australia: Sydney). Observations at the night roost were restricted to a pond nearest the North Arm of the river although birds also used a second pond at this site (Fig.

3.2). Observations at the night roost were over a shorter period, from one hour before to one to two hours after published high tide.

I used instantaneous flock scans (Altmann 1974), at 30 minute intervals, to record behaviour and microhabitat use of each Eastern Curlew at the roost (Table 3.2) onto a microcassette tape and transcribed later. The direction of each flock scan was randomised. Total counts of Eastern Curlews were recorded before each flock scan. Scans where birds were disturbed or departed from the roost were discarded from the analysis. Roosting behaviour was categorised into mutually exclusive behaviours: resting (head tucked in rear scapulars) (Handel & Gill 1992); preening (including bathing, scratching and coughing up digested pellets); vigilant (standing inactively; or head and body upright, scanning from side to side); foraging (probing substrate or handling prey item); walking/running; aggression (chasing or pecking at other birds); and flying.

Microhabitat types were site specific. At the rock wall, they included open water, rock surface and grass and at the sandspit, they included open water, sand flat, saltmarsh and grass/shingle. Microhabitats at the night roost were separated into open water, saltmarsh and mangrove/open water (where birds were within two metres of the mangrove edge). The total area (ha) of each microhabitat was estimated from aerial photography of the estuary using a Geographic Information System (GIS) (ArcView 3.2). Open water available to birds at the roost sites was the maximum wading depth of Eastern Curlews, which is about 128 mm (± 2.3 SE) (see Chapter 4). This represented a strip about 0.5 m wide adjacent to the mean high water mark on the rock wall and at the sandspit deep sections in the lagoon (> 128 mm) were excluded.

3.3.3 Focal observations

I used focal animal sampling (Altmann 1974) to investigate resting, vigilance and sleeping behaviour in individual birds at the roost sites (Table 3.2), recorded into a micro-cassette tape recorder over a two minute period. Birds were selected randomly and the initial behaviour, microhabitat, distance (relative to bird length) and species of nearest neighbour recorded. Focal samples were aborted if birds were disturbed and were not restarted until 15 minutes later to allow birds to resume normal roosting behaviour (Sheater & Burger 1992). Observations were transcribed later to determine the proportion of time birds spent resting, vigilant and sleeping at the roosts. Sleep behaviour was only recorded for focal birds at the sandspit (n = 76) and the

night roost (n = 49) (Feb-Mar 2006) but not at the rock wall because of the movement of the boat. I used a telescope to observe eye movements in birds at the sandspit. At night the infra-red spotlight reflected on open eyes of focal birds permitting observations of sleeping behaviour. The spotlight did not appear to alter the behaviour of birds during flock scans or extended focal sample observations. Sleep was measured by recording change in eye movement from opened to closed, with 'peeking', any opening of the eyelid, representing vigilance behaviour (Dominguez 2003). The proportion of time birds spent with their eye closed was assumed to represent the amount of time birds were asleep (Amlaner & McFarland 1981).

3.3.4 Disturbance

I recorded disturbance events at the day roosts (Oct-Dec 2005; Feb-Mar 2006) but I could not quantify disturbance at the night roost. At the day roosts, disturbances were events that caused birds to take flight. I estimated the proportion of flock that took flight, the cause and the total time birds spent in flight onto a microcassette recorder. The disturbance time (secs) represented the total time from when birds first took flight to the last bird landing back at the roost. During each survey of the rock wall, I also noted any boats passing in the river channel to measure background activity at this site.

3.3.5 Statistical analyses

Flock scans have problems of non-independence, as the position and behaviour of individual birds can depend partly on conspecifics in the same flock. To be conservative I summed behavioural data across flock scans and concentrated on dominant behaviours that represented more than 10% of bird activity (resting and vigilance). I used Analysis of Covariance (ANCOVA) to investigate differences in flock behaviour at the day roosts between seasons: October – December 2005 (non-migratory season) and February – March 2006 (pre-migratory season). Flock size was used as a covariate in this seasonal analysis as it is known to influence shorebird behaviour (Whitfield 2003a; Yasue 2005). Following inspection of the data for normality and homoscedasticity, I transformed the behaviour percentage data using the arcsine transformation (Fowler *et al.* 1998). Due to non-normal distributions, I used non-parametric Kruskal-Wallis tests, comparing the *H*-statistic to chi-square distribution, to investigate differences in flock behaviour and microhabitat use among the roost sites and to determine the effect of flock size on resting and vigilance behaviour (Quinn & Keough 2002). For the flock size analysis, I categorised flock sizes as either small ($\leq$ 20 birds), medium (> 20 - 100 birds) or

large (> 100 birds). To account for the relative availability of microhabitats at the roost sites, I calculated ratios of usage (proportion of birds in each microhabitat) to availability of each microhabitat (the proportion of each microhabitat at the roost sites) (Johnson 1980). I used Kruskal-Wallis tests to investigate differences in resting and vigilance times in focal birds across three tidal periods: before (-2 hours to 0 hours); high (0 hours to +2 hours); and after (+2 hours to +4 hours) (where 0 = published high tide). I only compared observations recorded in the 'before' and 'high' roosting periods at the night roost as only four focal observations were completed in the 'after' period. I used a One-way Analysis of Variance (ANOVA) to determine whether nearest-neighbour distances, following log-transformation (log10 (x+1)), differed among the roost sites for focal birds. I used ANCOVA tests to determine if sleep behaviour differed between the sandspit day roost (n = 76) and night roost (n = 47) between February – March 2006. Log-transformed nearest-neighbour distances were a covariate in this analysis. I used two measures to investigate sleep behaviour (Dominguez 2003): (i) peeking time (the total time (%) a bird had its eye open in the focal sample) (transformed with arcsine); and (ii) peeking rate (number of eye openings per minute) (transformed with square-root). I used a two-way ANOVA to determine whether log-transformed disturbance rates and total disturbance time differed between the day roosts or weekend and weekday surveys. All data were analysed with SPSS (2005). Means and standard errors (SE) are presented throughout.

3.4 Results

3.4.1 Flock behaviour

Based on the HBOC's monthly surveys of the estuary, most Eastern Curlews roosted at the sandspit (67.6 ± 16.7%) and rock wall (30.3 ± 17.3%) during day-time high tides, while only a few birds roosted in ponds on Ash Island (2.1 ± 0.6%) and Fullerton Cove beach (0.1 ± 0.1%) (Oct 2005 – Mar 2006). Generally, flock behaviour was similar at the three major roost sites (Fig. 3.3). There was no difference in the proportion of resting (X^2 = 0.5, df = 2, p = 0.785) or vigilant (X^2 = 4.0, df = 2, p = 0.138) birds among the roost sites. Mean flock size ranged from 48.8 ± 2.6 birds at the rock wall, 96.2 ± 6.6 birds at the sandspit to 10.4 ± 1.3 birds at the night roost, however, flock size was not related to levels of resting (X^2 = 3.5, df = 2, p = 0.177) or vigilance (X^2 = 1.0, df = 2, p = 0.616) behaviour at the roost sites. Resting behaviour formed over half of flock activity at the rock wall (50.1 ± 2.5 %), sandspit (52.1 ± 2.1%) and night roost (54.8 ± 4.9%) (Fig. 3.3). Vigilance behaviour was the next most common behaviour (31.9 – 41.2%) followed by preening (5.4 – 6.6%) and walking/running (1.6 – 5.0%). Less than 2.5% of the flock foraged at the roost sites. Few Eastern Curlews were observed in flight (0 – 1.2%) or in aggressive encounters (0 – 0.2%) at the day roosts and no flight or aggressive behaviour was observed during flock scans at the night roost (Fig. 3.3). The proportion of resting Eastern Curlew during flock scans of the day roosts did not differ between seasons (rock wall: $F_{1,205}$ = 0.9, p = 0.346; sandspit: $F_{1,214}$ = 2.5, p = 0.116), however, a higher proportion of birds were vigilant during flock scans in February – March 2006 (pre-migratory) compared to October – December 2005 (non-migratory) at both the rock wall ($F_{1,205}$ = 5.6, p = 0.019) and sandspit ($F_{1,214}$ = 4.3, p = 0.017) (Fig. 3.4).

Microhabitat use was site specific. Most birds were observed on the rock surface (96.5 ± 0.5%) at the rock wall, while few birds used open water (1.5 ± 0.3%) or grass on the upper surface of the rock wall (2.1 ± 0.4%). The ratios of usage-availability confirmed this habitat preference (X^2 = 364.8, df = 2, p < 0.001). At the sandspit, more birds were observed in open water compared to areas of sandflat, saltmarsh and grass/shingle (X^2 = 137.1, df = 3, p < 0.001), but birds also showed some positive association for areas of sandflat (Table 3.3). Eastern Curlews also showed some preference to roost in open water at the night roost (X^2 = 81.9, df = 2, p < 0.001) but did not avoid the mangrove edge (Table 3.3).

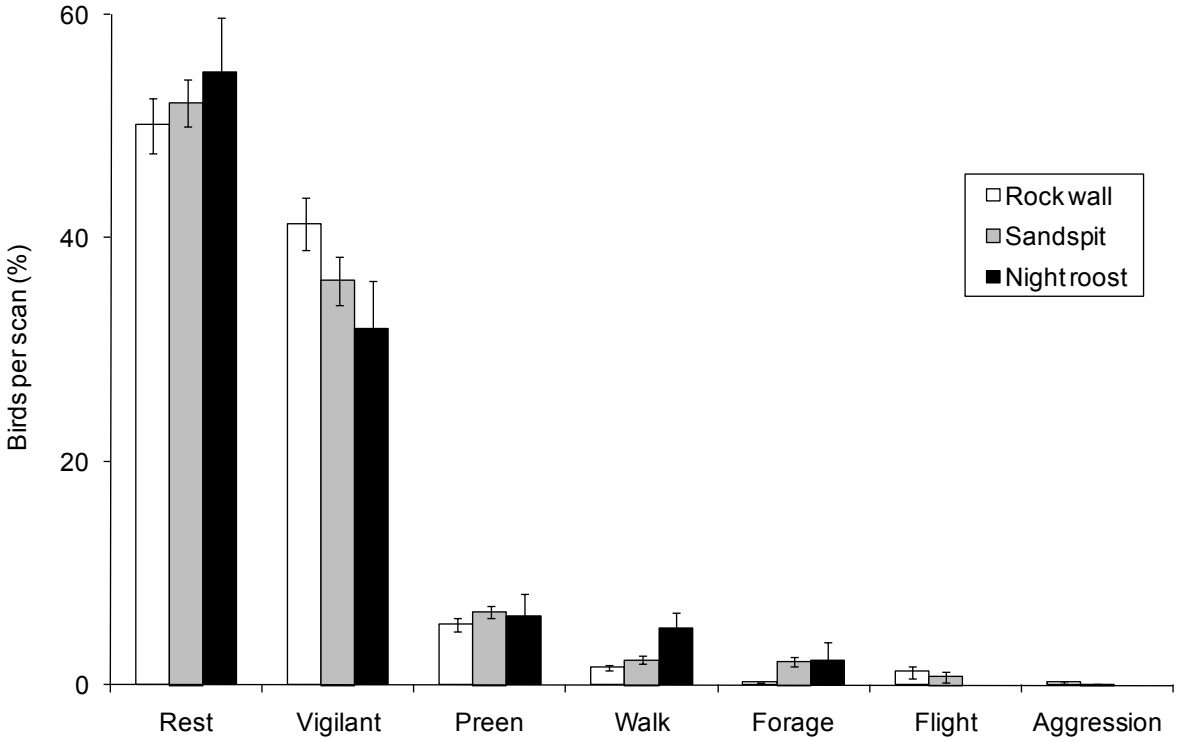


Figure 3.3 Mean (± S.E.) percentage of behaviours observed in Eastern Curlew flocks roosting at the rock wall, sandspit and night roost sites.

Table 3.3 Total area of each microhabitat and their use by Eastern Curlews at the roost sites.

Roost	n	Microhabitat	Area[a] (ha)	Usage[b] (%)	Selection[c] index
Rock wall	208	open water	0.04	1.5 (0.3)	0.2 (0.1)
		rock	0.38	96.5 (0.5)	1.2 (0.0)
		grass	0.06	2.1 (0.4)	0.3 (0.0)
Sand spit	221	open water	0.24	32.3 (2.2)	3.0 (0.2)
		sand flat	0.44	24.3 (1.9)	1.2 (0.1)
		saltmarsh	1.10	28.0 (1.9)	0.6 (0.0)
		grass/shingle	0.46	15.5 (2.0)	0.7 (0.1)
Night roost	51	open water	0.53	88.0 (3.1)	1.4 (0.1)
		saltmarsh	0.29	5.6 (1.9)	0.2 (0.1)
		open water/mangrove	0.04	6.2 (2.6)	1.3 (0.6)

[a] Mapped from aerial photographs.
[b] Mean (± S.E.) percentage of birds observed in each microhabitat.
[c] Proportion of observations of birds in each microhabitat as a proportion of microhabitat available (values >1 indicate greater association) (see Methods).
n = number of flock scans.

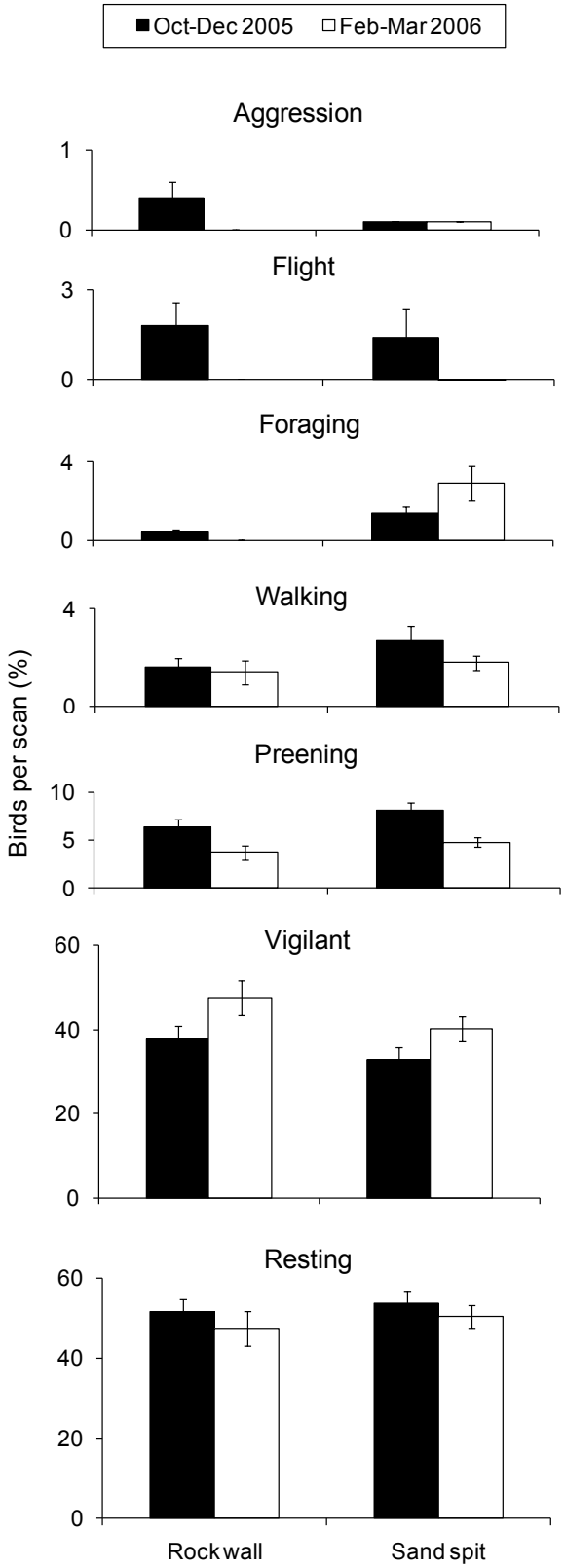


Figure 3.4 Seasonal patterns in Eastern Curlew roosting behaviour at the day roosts.

3.4.2 Focal birds

The amount of time focal birds spent resting at the rock wall increased in later stages of the tidal period. Birds spent more time resting in the after period (69.5 ± 4.2%; n = 97) compared to the high (60.2 ± 5.1%; n = 83) or before (44.5 ± 5.4%; n = 73) tidal periods (X^2 = 13.5, df = 2, p = 0.001). This corresponded with reduced levels of vigilance over the roosting period at the rock wall (X^2 = 14.2, df = 2, p = 0.001) (Fig. 3.5). Resting and vigilance time in focal birds did not differ across the tidal period at the sandspit (resting: X^2= 0.5, df = 2, p = 0.761; vigilance: X^2 = 0.3, df = 2, p = 0.851). The proportion of time birds spent resting was similar in the after (56.6 ± 2.7%; n = 95), high (61.6 ± 2.5%; n = 129) and before (60.7 ± 2.4%; 141) roosting periods at the sandspit (Fig. 3.5). During observations at the night roost, there was no significant difference in the time birds spent in a resting (X^2 = 1.9, df = 1, p = 0.173) or vigilant (X^2 = 1.8, df = 1, p = 0.182) (Fig. 3.5) posture.

Nearest neighbour distances were similar at the day roosts (rock wall: 3.1 ± 0.4 bird lengths; sandspit: 3.1 ± 0.3 bird lengths), but greater at the night roost (4.8 ± 0.5 bird lengths) ($F_{2, 693}$ = 13.5, p < 0.001). Eastern Curlews were neighbours of 76%, 96% and 55% of focal birds at the rock wall, sandspit and night roost respectively. Although sleeping birds were spaced more widely at the night roost (4.1 ± 0.4 bird lengths) than at the sandspit day roost (2.6 ± 0.2 bird lengths), this did not significantly affect peeking behaviour. Overall, resting birds spent less time peeking at the sandspit (53.9 ± 2.0%) than at the night roost (69.1 ± 3.1%) during focal observations. However, resting birds peeked almost twice as often at the sandspit (14.4 ± 0.5 min^{-1}) compared to birds observed at the night roost (7.3 ± 0.5 min^{-1}) (Table 3.4).

Table 3.4 Results of ANCOVA of peeking time and peeking rate for Eastern Curlews in relation to roost site and nearest-neighbour distance.

Sources of variation		Peeking time (%)		Peeking rate (min^{-1})	
	df	*F*	*p*	*F*	*P*
Corrected	2	3.0	0.056	43.3	<0.001
Intercept	1	48.5	<0.001	482.5	<0.001
Nearest neighbour distance	1	2.0	0.163	0.6	0.448
Roost site	1	5.1	0.025	84.1	<0.001
Error	121				
Total	124				

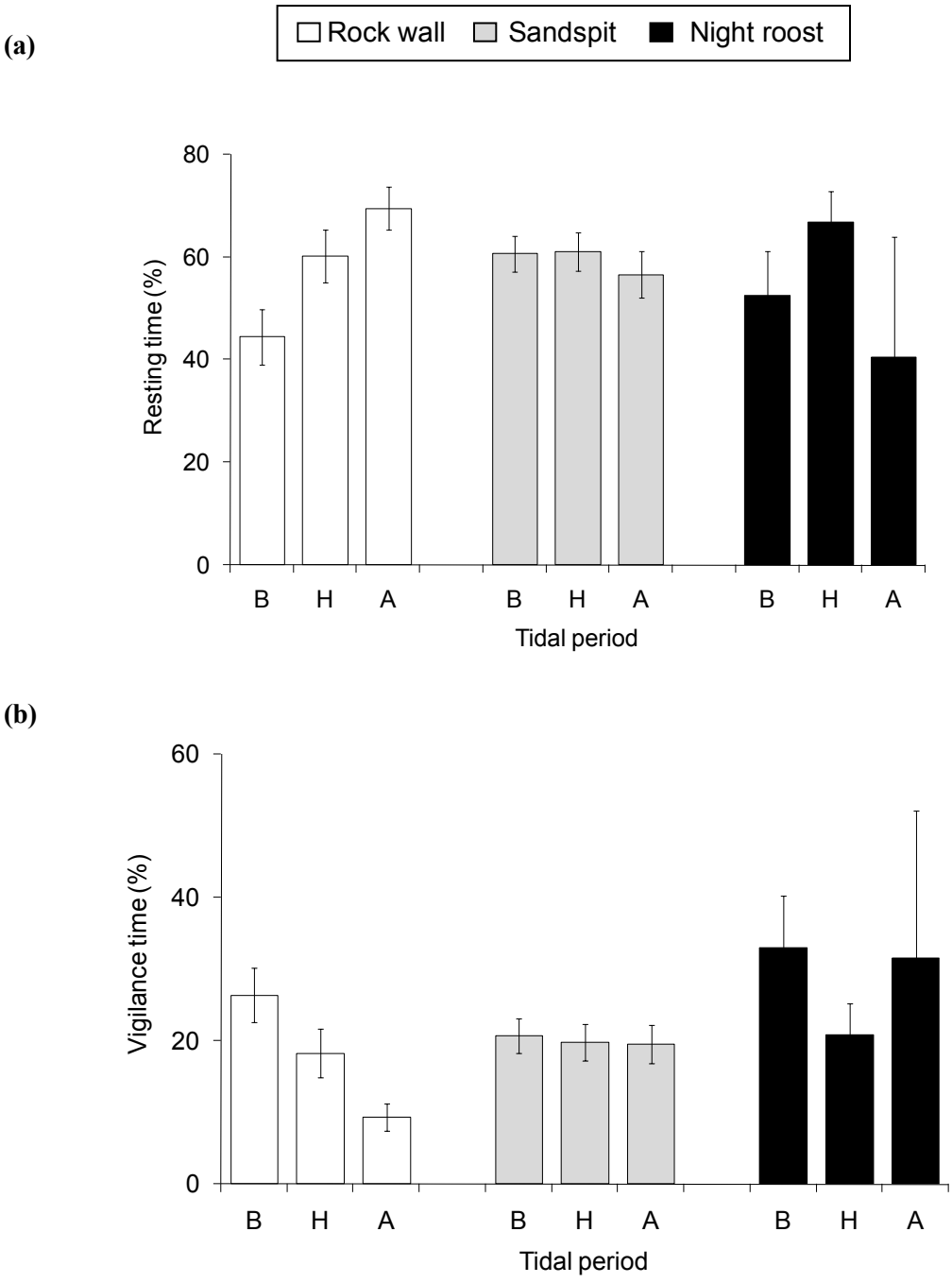


Figure 3.5 Mean (± S.E.) (a) resting and (b) vigilance time (%) spent by focal Eastern Curlews at the roost sites. The roosting period was divided into before (B) (-2 hours to 0), high (H) (0 to +2 hours) and after (A) (+2 to +4 hours) survey periods (where 0 = published high tide). Note that only four focal samples were recorded during the after period for the night roost and were excluded from the analysis.

3.4.3 Disturbance

Overall, disturbance rates were higher at the rock wall (1.7 ± 0.3 hr^{-1}) than at the sandspit (0.8 ± 0.1 hr^{-1}) ($F_{1,28} = 5.0$, $p = 0.034$) but did not differ significantly between weekends and weekdays ($F_{1,28} = 2.2$, $p = 0.151$) (Table 3.5). Birds of prey were the most frequent cause of disturbance at the rock wall (37 events). Drifting boats (28 events), passing boats (17 events), unknown causes (15 events) and wake from large cabin cruisers (13 events) were also common causes of disturbance at the rock wall. Fishers walked onto the rock wall causing birds to take flight on three occasions. Single disturbance events included a low flying helicopter, lightning and a raven (which was probably mistaken for a bird of prey). At the sandspit, the cause of many disturbances was unknown (26 events) but birds of prey were also a common cause (23 events) for birds to take flight. Other disturbances included bird watchers, walkers, fishers collecting weed from the lagoon for bait or live bait from sandflats in front of the lagoon (Table 3.5). Birds of prey which were responsible for disturbing roosting Eastern Curlews at the day roosts included: White-bellied Sea Eagles *Haliaeetus leucogaster*, Whistling Kites *Haliastur sphenurus*, Peregine Falcons and Swamp Harriers *Circus approximans*.

Table 3.5 Causes of disturbance and mean (± S.E.) percentage of flock taking flight and time spent in flight during 16 surveys of the rock wall and sandspit (Oct-Dec 2005; Feb-Mar 2006).

Disturbance type	Total events		Time in flight (secs)		% Flock in flight	
	Rock wall	Sandspit	Rock wall	Sandspit	Rock wall	Sandspit
Bird of prey	37	23	88.7 (9.0)	76.5 (11.1)	63.0 (6.7)	90.0 (5.4)
Boat drifting	28	-	84.5 (13.8)	-	40.4 (6.6)	-
Boat passing	17	-	75.1 (16.5)	-	39.7 (9.2)	-
Unknown	15	26	87.6 (20.7)	68.3 (13.1)	50.6 (12.6)	76.4 (7.1)
Boat wake	13	-	73.3 (11.8)	-	27.7 (8.7)	-
Bird watchers	-	8	-	67.3 (10.2)	-	73.1 (13.7)
Fisher	3	4	140.0 (93.9)	105.0 (0.0)	30.0 (22.5)	72.5 (13.1)
Fisher - weed collecting	-	4	-	85.3 (24.8)	-	100 (0.0)
Walker	-	4	-	118.3 (32.5)	-	77.5 (22.5)
Fisher/bird of prey	1	1	*	50.0 (-)	10.0 (-)	100.0 (-)
Helicopter	1	-	77.0 (-)	-	10.0 (-)	-
Lightning	1	-	*	-	75.0 (-)	-
Raven	1	1	*	20.0 (-)	5.0 (-)	50.0 (-)
Total	117	71	84.5 (6.4)	75.1 (6.4)	45.9 (3.7)	81.6 (3.8)

*Note that there were three occasions where total time of flight was not recorded accurately at the rock wall.

Most of the people-induced disturbances at the rock wall were caused by small recreational fishing boats (Table 3.5). The total number of boats passing the rock wall over the roosting period was high (5.3 ± 1.0 passes hr^{-1}), but boat traffic was higher at weekends (8.7 ± 1.5 passes hr^{-1}) than on weekdays (3.8 ± 1.2 passes hr^{-1}). This was part of background noise at the rock wall and on most occasions birds did not flush in response to passing boats. However, birds were more likely to flush during spring high tides or when boats were large, or passed inside navigational markers (< 40m from rock wall), as this caused waves to cover the rock wall, displacing roosting birds. The proportion of disturbances caused by people was high at the sandspit on weekend (56.7 ± 19.4%) and weekday (24.3 ± 9.3%) surveys. Despite disturbance rates being higher at the rock wall, there was no occasion when the entire flock deserted the roost in favour of the sandspit. In contrast, there were nine occasions where the whole flock was forced to leave the sandspit when disturbed and relocate to the rock wall. Furthermore, a smaller proportion of the flock took flight at the rock wall (45.9 ± 3.7%) than at the sandspit (81.6 ± 3.8%) following a disturbance (Table 3.5).

Overall, birds spent a similar amount of time in flight after being disturbed at the rock wall (84.5 ± 6.4 secs) and sandspit (75.1 ± 6.4 secs) ($F_{1,160} = 0.6$, $p = 0.429$) (Table 3.5). The maximum total time birds spent in flight over a high tide period (as a result of human, natural and unknown disturbances) was high at both the rock wall (24 mins 51 secs) and the sandspit (11 mins 51 secs). At the rock wall, Eastern Curlews spent longest in flight after being disturbed by fishers on foot (140.0 ± 93.9 secs); birds of prey (88.7 ± 9.0 secs) and drifting boats (84.5 ± 13.8 secs). At the sandspit, walkers (118.3 ± 32.5 secs) and fishers (105.0 ± 0.0 secs) caused birds to stay in flight longest, but birds of prey (76.5 ± 11.1 secs) and fishers collecting weed (85.3 ± 24.8 secs) also forced birds to stay in flight for extended periods of time (Table 3.5).

3.5 Discussion

Eastern Curlews spent over 50% of their time at the roost sites in a resting posture, which is consistent with other studies of shorebird roosting behaviour (Handel & Gill 1992; Rogers 2003). A large proportion of the flock (30 - 40%) remained vigilant at roosts over the high tide period (Fig. 3.3). These patterns were evident in flock and focal bird behaviour, with each bird balancing the need for rest with scanning for potential predators (Lendrem 1983). Eastern Curlews also became more agitated immediately prior to their migration, with vigilance levels increasing in late stages of their non-breeding season (Fig. 3.4).

Estuary-wide surveys by the Hunter Bird Observers Club, confirmed that the rock wall and sandspit provided the most important day roosting habitats for Eastern Curlews in the Hunter estuary, but birds of prey, recreational boats, walkers and fishers were significant causes of disturbance at these sites. A high proportion of birds was flushed from the sandspit after a disturbance and Eastern Curlews would often desert this site after a single disturbance event. It may be that the sandspit was suboptimal, as levels of resting time remained the same over the roosting period in focal birds at the sandspit, but birds rested for longer with more time spent at the rock wall. The most common cause for birds to be disturbed at the sandspit was unknown (26 events), which suggests that the birds were more 'flighty' at this site. Shorebirds adopt risk aversion behaviour and will often take flight as a result of false alarms (Handel & Gill 1992; Burton *et al.* 1996). Vigilance levels also decreased at the rock wall over time, with lowest levels of vigilance recorded at the end of the roosting period, but vigilance remained constant over the roosting period at the sandspit (Fig. 3.5). This could have been an effect of high levels of background noise from vehicles crossing Stockton Bridge, which is located above the sandspit (Fig. 3.2). Disturbance rates, however, were higher at the rock wall and this site had fewer microhabitat types available compared to the sandspit.

Birds at the sandspit were observed lying down on wet sandflats during high ambient temperatures (> 25ºC) and this may have been in an attempt to keep cool as there is mounting evidence that shorebirds are susceptible to heat stress in their non-breeding sites (Battley *et al.* 2003; Rogers *et al.* 2006b). Open water was also important for birds roosting at night. Shorebirds may depend on the noise created by a predator moving through the water as a form of predator detection. Roosting birds were spaced more widely at the night roost, where they may

need more space to take flight in the event of predator attack. The alternative explanation could be that it is too risky for birds to be widely spaced during the day because of the threat of attack by birds of prey but this is not an issue for birds at night.

Peeking time (vigilance) was higher in focal birds at night, but birds peeked less frequently than during the day, suggesting that Eastern Curlews may sleep more deeply at night. At night, birds may also spend more time scanning for ground predators, as shorebirds generally have poor night vision (Rojas *et al.* 1999; Rogers *et al.* 2006b). Sleep is thought to be a physiological necessity for most animals (Rattenborg *et al.* 1999). Great Knots *Calidris tenuirostris* in Roebuck Bay, north-western Australia, for example, spent on average 50 minutes in a resting posture during the day but up to five hours resting at night (Rogers 2003). These differences were thought to be related to perceived levels of predation risk, with there being greater advantages to sleeping more at night and being more vigilant for diurnal birds of prey during the day (Rogers 2003; Rogers *et al.* 2006b).

Although only half of the night roost was studied, relatively small flocks (10.4 ± 1.3 birds) of Eastern Curlews were observed. Given that the Hunter estuary regularly supports about 400 Eastern Curlews during summer months (Sept-Apr), this suggests that there are additional night roosts in the estuary which have not been located. By spreading out over multiple roosts at night shorebirds may be less conspicuous to ground predators. During opportunistic night surveys in other parts of the estuary, flocks of five or more Eastern Curlews were also observed roosting on Milhams Pond and Wader Pond on Ash Island, on the western side of Kooragang Island (Fig. 3.1). These areas are similar to the Windeyers Reach night roost, with shallow open ponds fringed by mangrove and low level saltmarsh vegetation. In contrast, no migratory shorebirds were observed at the rock wall during night surveys and only single Bar-tailed Godwits and Eastern Curlews were observed at the sandspit. These sites would be unattractive to shorebirds at night, as both roosts can be accessed by ground predators and are illuminated by artificial lighting from Stockton Bridge and industrial machinery on Kooragang Island, making shorebirds visible to ground predators, such as foxes, which have been observed at both day roosts and on Ash Island (Richardson 2004; J. Spencer *pers. obs.*).

Roosts originally found along the foreshore of the main channels and islands in the Hunter estuary were removed by major dredging during port development. Today, much of the available roosting habitat is artificial and is threatened by high levels of human-induced disturbance and by further port development, erosion, mangrove encroachment and eustatic sea level rise (Howe 2008). Recreational boats and fishers were a significant cause of disturbance of Eastern Curlews in this study and of Bar-tailed Godwits *Limosa lapponica* in the Hunter estuary in the summer of 2004-05 (Foate 2005). Disturbance was most intense during spring high tides when the availability of roosting habitat was most limiting in the estuary. This level of disturbance may have implications for the viability of major day roosting habitat in the Hunter estuary. Cumulative disturbance events can impinge on fat reserves needed for migration and moult in migratory shorebirds (Burger & Gochfeld 1991) and, in some cases, high levels of natural or anthropogenic disturbance can cause roost abandonment (Dekker 1998; Hotker 2000; Colwell *et al.* 2003). Many shorebirds are affected by disturbance but larger 'flighty' species, such as Eastern Curlews, Whimbrels *Numenius phaeopus* and Common Greenshanks *Tringa nebularia* with longer 'critical' flight distances may be most affected (Lawler 1994; de Boer & Longamane 1996; Blumstein *et al.* 2003). Over a single high tide period, Eastern Curlews in the Hunter estuary could spend up to 20 minutes in flight as a result of human-induced disturbance. In comparison, increases in daily energetic costs of the equivalent of ten or more minutes extra flying time a day had significant impacts on the body condition and mortality rates of shorebird species in a northern hemisphere study (Durell *et al.* 2005). Although admittedly, the costs of energy conservation would be greater in colder climates.

Protecting roosting habitat should be a priority for conserving shorebird populations in the Hunter estuary. Most roost sites hold stable populations of shorebirds, with some migratory species showing high site fidelity between and within their non-breeding seasons (Dann 1981a; Rehfisch *et al.* 1996). Fullerton Cove provides rich food supplies for shorebirds in the Hunter estuary (Hutchings 1977, 1983; Geering 1995) (see Chapter 5) which attracts returning migratory shorebirds each year, but good quality high tide roosts are equally important. For a feeding area to be viable, it is essential to provide multiple roost sites within close proximity so that shorebirds can use alternative roosting habitat in the event of disturbance (Dias *et al.* 2006; Rogers *et al.* 2006b). Rehabilitation works have already been undertaken to improve roosting habitat in the Hunter estuary, however, further steps are required to manage public access to the day roosts and

to prevent mangrove encroachment into saltmarsh areas. Further research is also required to locate remaining night roosts in the estuary, so that the management of these habitats can be prioritised accordingly.

CHAPTER 4: THE IMPORTANCE OF MUDFLAT IMPOUNDMENTS FOR MIGRATORY SHOREBIRDS IN THE HUNTER ESTUARY, SOUTH-EASTERN AUSTRALIA

4.1 Abstract

I investigated the importance of artificially-impounded intertidal mudflats for migratory shorebird species in the Hunter estuary, south-eastern Australia (Oct-Dec 2004). These mudflats are exposed for longer than low tide feeding habitat available in the rest of the Hunter estuary. Fifteen migratory shorebird species were recorded in the survey area. Bird density could be high in the largest impoundment (47 ± 4 birds ha^{-1}), with counts of five species representing over 65% of their populations in the Hunter estuary during the 2004-05 non-breeding season. Tide type significantly affected the total number of birds using the artificial impoundments, with greater numbers observed during neap tides when feeding habitat was most limiting in the rest of the estuary. The impoundment provided important foraging habitat for Black-tailed Godwits *Limosa limosa*, Curlew Sandpipers *Calidris ferruginea* and Red Knots *Calidris canutus* and was also an important staging roost for larger shorebird species, such as Eastern Curlews *Numenius madagascariensis* and Bar-tailed Godwits *Limosa lapponica*. Behavioural activity varied with the tidal period, with fewer birds feeding closer to high tide, however, many small shorebirds would feed almost continuously over the exposure period until they were forced to relocate to nearby high tide roosts. Although these mudflat impoundments operate artificially, they play a significant role in maintaining migratory shorebird species in the Hunter estuary. There needs to be greater recognition of the conservation importance of these artificial mudflats in future management of estuarine habitats in this non-breeding site.

4.2 Introduction

Coastal wetlands support large numbers of migratory shorebirds during their non-breeding season (Lane 1987; van de Kam *et al.* 2004). Most shorebird species forage on intertidal mudflats during low tide and are forced to move to high water roosts as the tide advances. The cycle of mudflat inundation and exposure affects the availability of invertebrate prey (Pienkowski 1983), causing shorebirds to move from one foraging area to another (Conners *et al.* 1981; Burger 1997). Diets and feeding habitats often overlap (Dann 1981b, 1987) but there is a segregation of shorebird species through a combination of prey preferences, foraging technique, sediment penetrability, bill length and water depth (Table 4.1) (Recher 1966; Burger *et al.* 1977). The amount of time shorebirds spend feeding depends on their daily energy requirements and the availability of food. Smaller species usually need to feed longer during the tidal cycle as they consume smaller prey than larger shorebirds (Dann 1987; Zwarts *et al.* 1990b). Some shorebirds capitalise on additional food sources when their low tide feeding habitat is unavailable by feeding in artificial habitats such as salt pans, fish ponds, dyked wetlands, pasture or cultivations (Velasquez 1992; Weber & Haig 1997; Twedt *et al.* 1998; Masero *et al.* 2000; Colwell & Mathis 2001; Ma *et al.* 2004).

Table 4.1 Ranges in height, bill and tarsus length and estimates of mean (± S.E.) wadeable depths for eight common migratory shorebird species in south-eastern Australia.

Common name	Height[a] (cm)	Bill length[a] (mm)	Tarsus length[a] (mm)	Wadeable depth[b] (mm) mean	n
Eastern Curlew	60 – 66	158.0 - 179.5	88.2 - 91.6	128.5 (2.3)	4
Black-tailed Godwit	40 – 44	72.9 - 85.8	62.4 - 66.4	104.8 (6.4)	8
Bar-tailed Godwit	37 – 39	80.1 - 108.0	52.9 - 57.8	79.6 (2.1)	10
Common Greenshank	30 – 35	50.0 - 55.2	55.1 - 60.1	89.0 (1.3)	10
Marsh Sandpiper	22 – 26	39.4 - 40.1	51.6 - 52.7	81.6 (2.3)	5
Curlew Sandpiper	18 – 23	35.6 - 41.0	28.4 - 30.6	47.0 (1.1)	10
Sharp-tailed Sandpiper	17 – 22	23.9 - 25.0	28.9 - 30.5	44.6 (0.6)	10
Red-necked Stint	13 – 16	17.1 - 18.7	19.7 - 20.3	28.1(0.6)	10

[a] Sources: Higgins and Davies (1996) and Marchant and Higgins (1993).
[b] Wadeable depths are based on measurements of total leg length of preserved skins from the Australian Museum, Sydney (n = number of specimens).

Coastal developments and human-induced disturbance have degraded many feeding habitats for shorebirds in their non-breeding range (Burton *et al.* 2002a; Durell *et al.* 2005). The Hunter estuary, south-eastern Australia, is an important non-breeding site in the East Asian-Australasian Flyway (EAAF) (Smith 1991; Watkins 1993) but the development of the industrial port city of Newcastle has caused significant losses of estuarine habitats (Kingsford & Ferster Levy 1997; Williams *et al.* 2000). Despite evidence for long-term declines in migratory shorebird numbers in the estuary, remaining shorebird habitats are threatened by further development to support a growing coal export industry and urban population (see Chapter 2). It is now critical to protect remaining natural feeding habitats and to identify important artificial habitats. In this chapter, I investigated the importance of artificially-impounded mudflats for migratory shorebird species in the Hunter estuary. I carried out boat-based surveys in the lower estuary to determine the relative importance of intertidal mudflats in the North Arm of the Hunter River and quantified migratory shorebird behaviour in a mudflat impounded by a large river training wall. This artificial mudflat was exposed for longer than low tide feeding habitat available in the rest of the Hunter estuary and was thought to provide supplementary feeding habitat for some shorebirds (Geering 1995).

4.3 Methods

4.3.1 Study site

This study was done in the Hunter estuary in New South Wales (NSW), south-eastern Australia (32° 51'S/ 151° 46'E) from October to December 2004 (20 days; 80 hours). Fullerton Cove is the preferred feeding site of many shorebird species in the Hunter estuary, containing about 750 ha of intertidal mudflat during spring low tides (Geering 1995). Shorebirds also forage on mudflats and sandflats surrounding Stockton sandspit, Sandy Island and Fern Bay (about 100 ha in total) and in mudflat impoundments (about 25 ha) on the western shoreline of the river (Figs. 4.1; 4.2) (more details on mudflat characteristics are in Chapter 5). The main day roosts are an artificial rock wall (Kooragang dykes) (Fig. 4.1) and Stockton sandspit (see Chapter 3). During their non-breeding season (Sept-Apr), about 85% of migratory shorebirds roost on the Kooragang dykes during day time high tides (Geering 1995; Kingsford & Ferster Levy 1997; Straw 1999). The Kooragang dykes are about 1.6 km in length and impound intertidal mudflats in four ponds (Fig. 4.1). These mudflats replaced the original mudflats and sandflats along the river foreshore, destroyed by dredging and reclamation in the early 1900s (see Chapter 2) (Kingsford & Ferster Levy 1997; Richardson 2004). Large sections of the Kooragang dykes have eroded since construction and breaks allow the impoundments to fill and drain with the tidal cycle, lagging behind the cycle in the main river channel by about two hours. Tide cycles are semidiurnal with usually one high tide during day and night. High tide heights ranged from 1.04 to 2.03 m and low tides from 0.13 to 0.79 m during this study (BOM 2004).

4.3.2 Boat surveys

I randomly stratified low tide surveys (Hoenig *et al.* 1993) (Nov – Dec 2004) of the lower estuary (Fig. 4.1) into neap (0.48 – 0.78 m; N = 5) and spring (0.13 – 0.20 m; N = 5) tide types. Fullerton Cove was not surveyed in this study as it was too shallow to be navigated by boat at low tide. Boat-based observations, with direction of travel and start point (Fig. 4.1) randomly allocated, were restricted to good weather conditions (wind speeds < 15 knots; sea states < 2 Beaufort scale) in the North Arm of the Hunter River (Fig. 4.1). Binoculars (8 x 30 mm) were used to identify and count migratory shorebird species, with Black-tailed Godwits *Limosa limosa* and Bar-tailed Godwits *Limosa lapponica* grouped as 'godwits' as they were difficult to distinguish from the boat. I accessed the Kooragang dykes on foot to observe birds in the mudflat impoundments with a telescope (Swarovski 20 – 60X zoom).

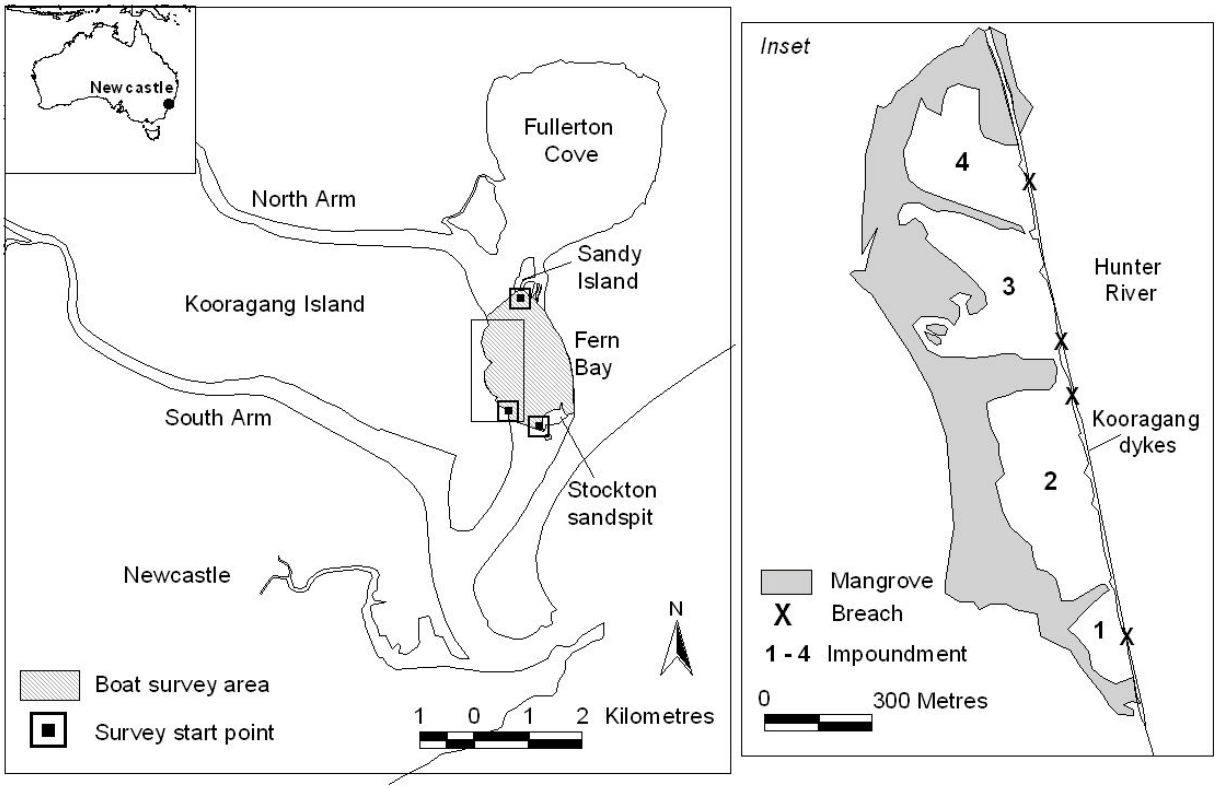


Figure 4.1 Location of the study site, the Hunter estuary, south-eastern Australia, the area for boat surveys, including the Kooragang dykes and associated mudflat impoundments (see inset). Detailed behavioural observations were restricted to impoundment number two.

Credit: Kooragang Wetland Rehabilitation Project (left), J. Spencer (right).

Figure 4.2 Aerial view of the Kooragang dyke system during a flood tide with major coal-loading facilities and wetlands on Kooragang Island in the background (left); ground view at Stockton sandspit at low tide looking north along the foreshore of Fern Bay towards Sandy Island (right).

4.3.3 Behaviour scans

Detailed observations of shorebird behaviour were restricted to impoundment two (Fig. 4.1), as this mudflat was largest (8.5 ha) and exposed for longest (Oct-Dec 2004; N = 20). The main impoundment was surveyed from low tide until about one to two hours before high tide when the incoming tide forced birds to move to the day roosts. Instantaneous flock scans (Altmann 1974) were done every 30 minutes (n = 137) to record flock behaviour and total numbers of each shorebird species. Direction of each flock scan was randomised and observations were recorded onto a tape recorder. Shorebird behaviour was categorised as: foraging (pecking/ probing substrate or handling prey item); resting (head tucked in rear scapulars); preening (including bathing, scratching and coughing up digested pellets); loafing (inactive); or other (walking/running, aggression and flying). Behavioural scans were discarded if flocks departed or were flushed by birds of prey. I sub-sampled large flocks (>300 birds) of Eastern Curlews *Numenius madagascariensis* and Bar-tailed Godwits by recording the behaviour of three subflocks (each > 50 birds) within each flock.

4.3.4 Data analysis

To investigate the distribution of shorebirds in the lower estuary, I grouped low tide counts of shorebird species as either large (body mass > 200 g: Eastern Curlews, Whimbrels *Numenius phaeopus*, Bar-tailed Godwits, Black-tailed Godwits and Grey Plovers *Pluvialis squatarola*), medium (100-200 g: Common Greenshanks *Tringa nebularia*, Pacific Golden Plovers *Pluvialis fulva*, Ruddy Turnstones *Arenaria interpres*, Great Knots *Calidris tenuirostris* and Red Knots *Calidris canutus*) or small (<100 g: Red-necked Stints *Calidris ruficollis*, Sharp-tailed Sandpipers *Calidris acuminata*, Curlew Sandpipers *Calidris ferruginea*, Marsh Sandpipers *Tringa stagnatilis* and Terek Sandpipers *Xenus cinereus*) (Marchant & Higgins 1993; Higgins & Davies 1996). I used a two-way Analysis of Variance (ANOVA) to investigate the effect of tide type and site (mudflat impoundments versus river foreshore) on the total number of migratory shorebirds. Counts were log-transformed (log10 (x+1)) (Fowler *et al.* 1998) to meet the assumptions of normality (Kolmogorov-Smirnov's) and equality of variance (Levenes) tests. Total numbers of migratory shorebirds and flock behaviour were investigated in relation to tide time (Burger *et al.* 1977). Tide time was measured in hours before published high tide. To determine the relative importance of the largest mudflat impoundment, I compared maximum counts from impoundment two in Oct-Dec 2004 to high tide roost counts recorded by the NSW National Parks

and Wildlife Service (NPWS) (N = 14) and Hunter Bird Observers Club (HBOC) (N = 6) in the 2004-05 non-breeding season (Oct-Mar). To investigate site-usage for individual species, I calculated an abundance index for six of the more common shorebird species (present in > 39% of flock scans): Eastern Curlew, Whimbrel, Bar-tailed Godwit, Black-tailed Godwit, Curlew Sandpiper and Red Knot. Each abundance index was calculated by summing all counts for every survey and converting flock scans to a percentage of the daily sum for each species (Burger *et al.* 1977). This procedure minimised the effect of large influxes of some species into the main impoundment for only short periods. I used non-parametric two sample Kolmogorov-Smirnov tests (Quinn & Keough 2002) to determine whether the total number of birds and total number of foraging birds differed significantly between neap and spring surveys of the main impoundment. I also used a linear regression analysis to determine whether tide time affected the percentage of birds foraging in the impoundment, after transformation with arcsine (Fowler *et al.* 1998), and a two-tailed Pearson's correlation (Quinn & Keough 2002) to investigate whether foraging behaviour could be explained by average non-breeding body masses of shorebirds (Marchant & Higgins 1993; Higgins & Davies 1996). As Terek Sandpipers and Grey Plovers were only seen during 1% of flock scans, they were excluded from this behavioural analysis. Statistical analyses were carried out in SPSS (2005). Means and standard errors (S.E.) are presented throughout.

4.4 Results

4.4.1 Tide type

Overall, 15 species of migratory shorebirds were recorded during low tide surveys of the Hunter estuary. During low tide boat surveys, more shorebirds were observed along the river foreshore than the mudflat impoundments ($F_{1,56}$ = 2.9, p = 0.033). Although tide type did not significantly affect the total number of migratory shorebirds in the whole survey area ($F_{1,56}$ = 0.01, p = 0.940), tide type did effect site usage, with greater numbers of shorebirds in the mudflat impoundments during neap low tides ($F_{1,56}$ = 6.3, p = 0.015) (Fig. 4.3).

Fourteen shorebird species were recorded in the main study area, impoundment two (Ruddy Turnstones were not observed here) (Table 4.2). Four migratory shorebird species were observed regularly (> 50% flock scans): Eastern Curlew (91%); Bar-tailed Godwit (82%); Black-tailed Godwit (68%); and Whimbrel (64%). Marsh Sandpipers, Red-necked Stints and Pacific Golden Plovers were less common (< 13% of scans). Terek Sandpipers and Grey Plovers were only recorded on single occasions. Maximum counts of Black-tailed Godwits (290 birds) in the main impoundment exceeded population estimates for the Hunter estuary in the 2004-05 non-breeding season. This impoundment could also support over 65% of the Hunter estuary's Eastern Curlews, Bar-tailed Godwits, Red Knots and Curlew Sandpipers in 2004-05 (Table 4.2).

Total numbers of migratory shorebirds in the main impoundment varied with the tidal period, with numbers peaking at about three hours before high tide (Fig. 4.4) when mudflat availability was limiting in the main feeding areas in Fullerton Cove and in the main channel of the Hunter River. Tide type significantly affected the total numbers of migratory shorebirds using the main impoundment (Z = 1.36, df = 1 , p = 0.050), with more birds recorded on exposed mudflats in this impoundment during neap tides, when shorebirds could access feeding habitat up to one hour before high tide (Fig. 4.4a). Total numbers of foraging birds did not differ significantly between neap and spring surveys (Z = 0.85, df = 1, p = 0.459) but shorebirds foraged for longer during neap cycles (Fig. 4.4b).

Table 4.2 Sighting frequency, and mean (± S.E.) and maximum counts of migratory shorebird species recorded in flock scans of the main study area, impoundment two (Oct-Dec 2004; n = 137), in relation to peak counts from the entire Hunter estuary.

Common name	Frequency (% scans)[a]	Mean	Maximum counts		% peak
			Mudflat impoundment	Hunter estuary[b]	Hunter count[c]
Eastern Curlew	91	146.9 (11.3)	454**	673**	67
Bar-tailed Godwit	82	160.6 (18.6)	810	1165	70
Black-tailed Godwit	68	77.2 (6.7)	290	225	129
Whimbrel	64	1.2 (0.1)	4	126*	3
Red Knot	50	27.8 (5.1)	390	560	70
Curlew Sandpiper	46	34.3 (6.1)	298	420	71
Common Greenshank	45	5.1 (1.0)	76	273*	28
Great Knot	34	0.7 (0.1)	4	7	57
Sharp-tailed Sandpiper	24	1.9 (0.5)	40	955	4
Marsh Sandpiper	12	0.9 (0.3)	32	124*	26
Red-necked Stint	11	0.7 (0.2)	23	115	20
Pacific Golden Plover	9	2.9 (1.2)	100*	239*	42
Grey Plover	1	0.01 (0.01)	2	4	50
Terek Sandpiper	1	0.01 (0.01)	1	38	3

[a] Percentage of flock scans when each species was recorded in impoundment two.
[b] Hunter estuary population estimates are based on maximum counts recorded during high tide surveys between Oct 2004 – Mar 2005 by the NPWS and HBOC (N = 20).
[c] Percentage of Hunter estuary population observed in impoundment two (based on maximum counts)
* nationally significant numbers (1% of Australian population) (DEH 2005)
** internationally significant numbers (1% of flyway population) (Bamford *et al.* 2008)
N = number of surveys, n = number of flock scans.

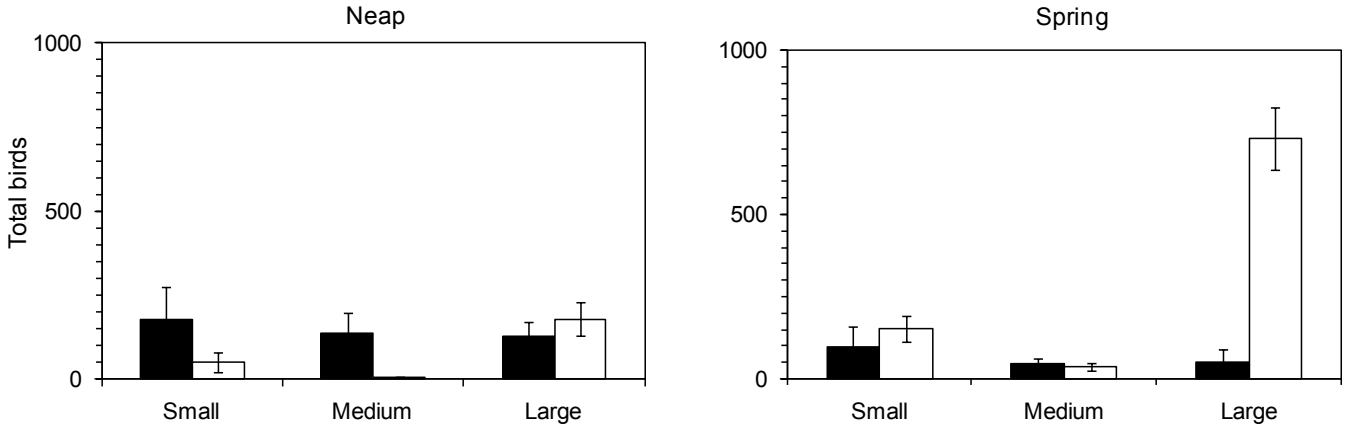


Figure 4.3 Mean (± S.E.) counts of small, medium and large migratory shorebird species recorded in all four mudflat impoundments (■) compared to the foreshore of the river channel (□) during neap (N = 5) and spring (N = 5) low tides surveys of the Hunter estuary (Nov-Dec 2004).

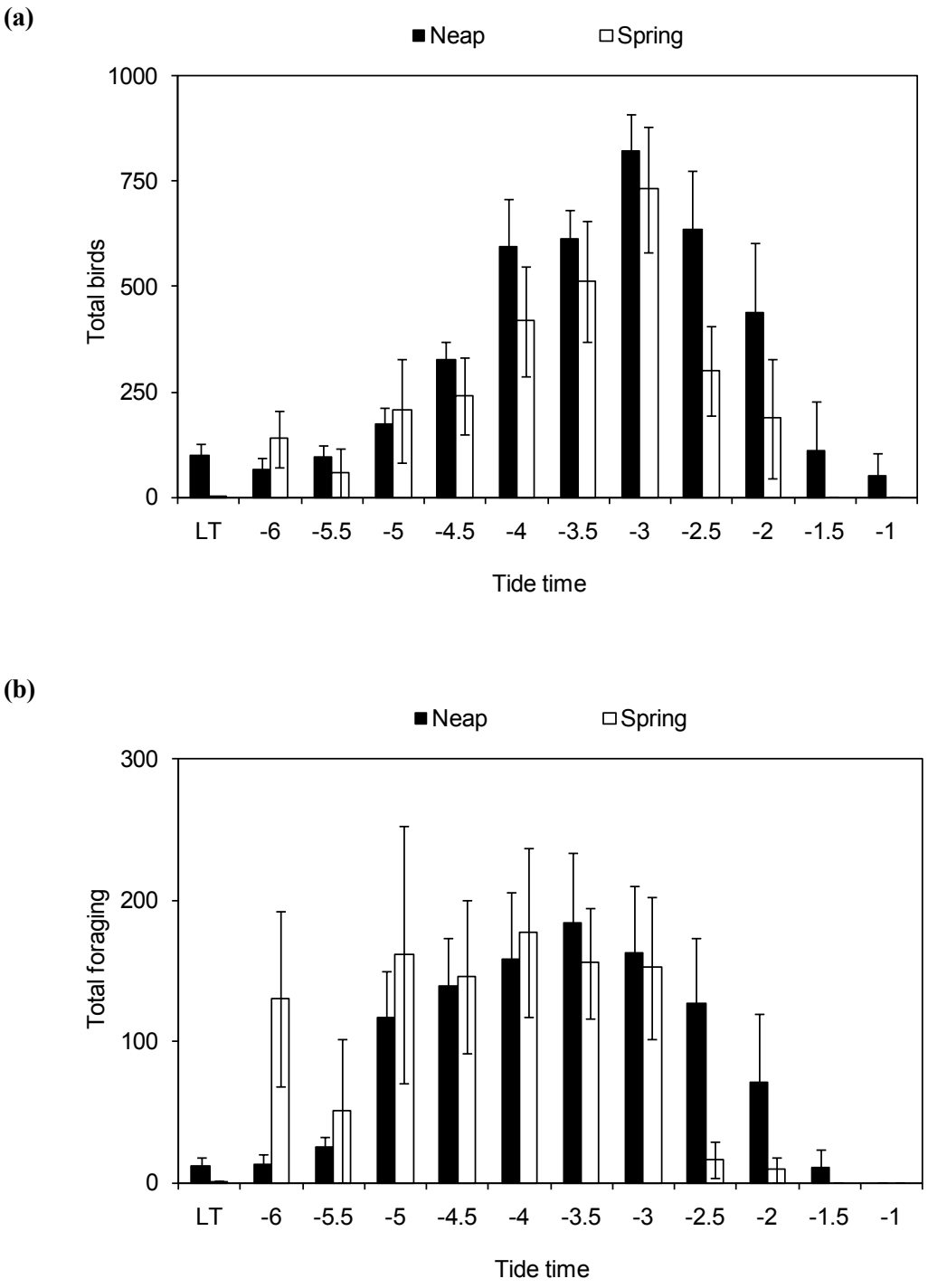


Figure 4.4 Mean (± S.E.) total (a) number of migratory shorebirds and (b) number foraging in the main impoundment during neap (N = 10) and spring (N = 10) low tides (Oct-Dec 2004). Tide time represents tide height measured in hours before published high tide (LT = low tide) (see Methods).

4.4.2 Flock behaviour

Dominant flock behaviour differed between shorebird species (Fig. 4.5). Many Eastern Curlews (50.0 ± 3.1%) and Bar-tailed Godwits (36.0 ± 3.4%) rested in the main impoundment, while Black-tailed Godwits (68.7 ± 4.5%), Common Greenshanks (80.6 ± 10.4%), Curlew Sandpipers (68.3 ± 4.8%), Great Knots (71.7 ± 6.2%), Marsh Sandpipers (93.3 ± 5.9%), Red Knots (82.8 ± 4.1%) and Red-necked Stints (97.7 ± 2.3%) predominantly foraged. Sharp-tailed Sandpipers and Pacific Golden Plovers were less active, with < 50% of birds foraging (Fig. 4.5). Bar-tailed Godwits and Red Knots were not recorded in the main impoundment at low tide but small numbers started arriving from five hours before high tide and larger flocks arrived at about three hours before high tide (Fig. 4.6). Overall, foraging behaviour varied with tide time, with fewer birds foraging closer to high tide ($r^2 = 0.21$, $F_{1,155} = 40.6$, $p < 0.001$) (Fig. 4.4). The proportion of foraging birds also declined over the survey period for the six most common shorebird species, but a high proportion of Black-tailed Godwits (35.2 ± 19.7%), Curlew Sandpipers (64.3 ± 17.8%) and Red Knots (52.2 ± 22.4%) continued to forage close to high tide (Fig. 4.6).

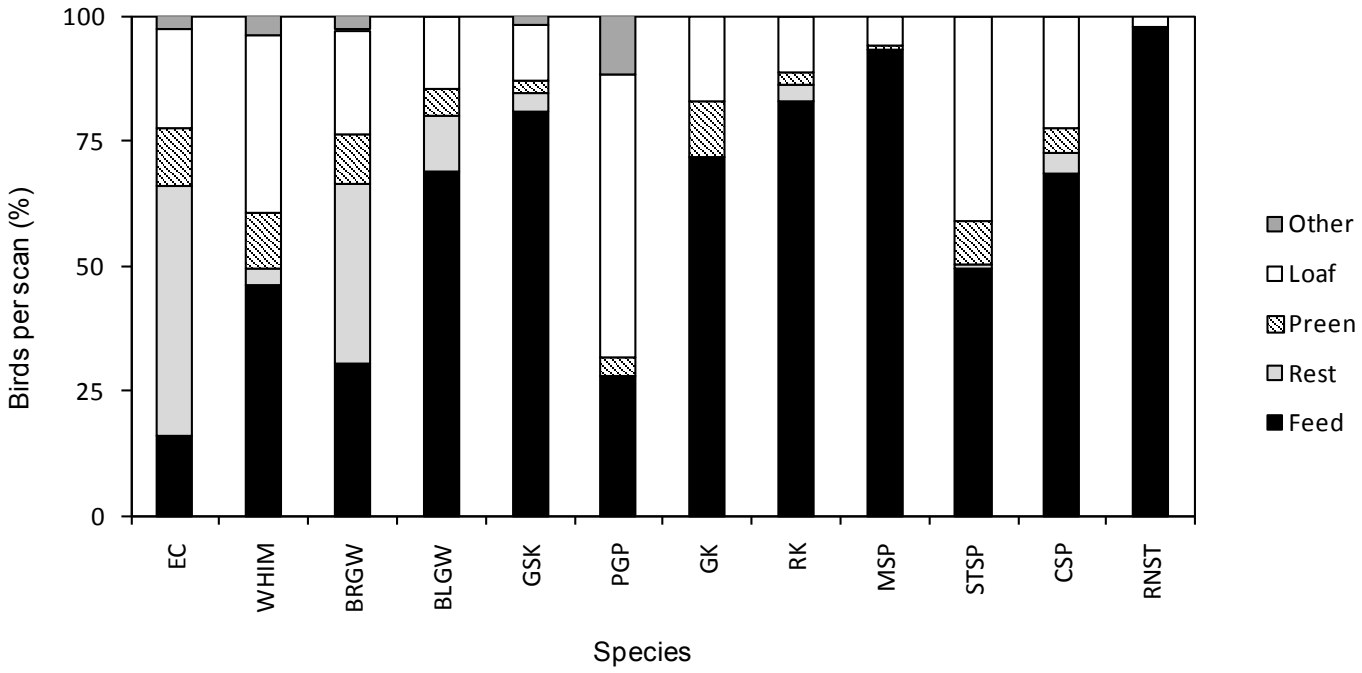


Figure 4.5 Overall flock behaviour for 12 migratory shorebird species in the main study area (impoundment two) in the Hunter estuary. The data are averaged over 20 survey days (Oct-Dec 2004). (Species codes: EC = Eastern Curlew, WHIM = Whimbrel, BRGW = Bar-tailed Godwit, BLGW = Black-tailed Godwit, GSK = Common Greenshank, PGP = Pacific Golden Plover, GK = Great Knot, RK = Red Knot, MSP = Marsh Sandpiper, STSP = Sharp-tailed Sandpiper, CSP = Curlew Sandpiper, RNST = Red-necked Stint).

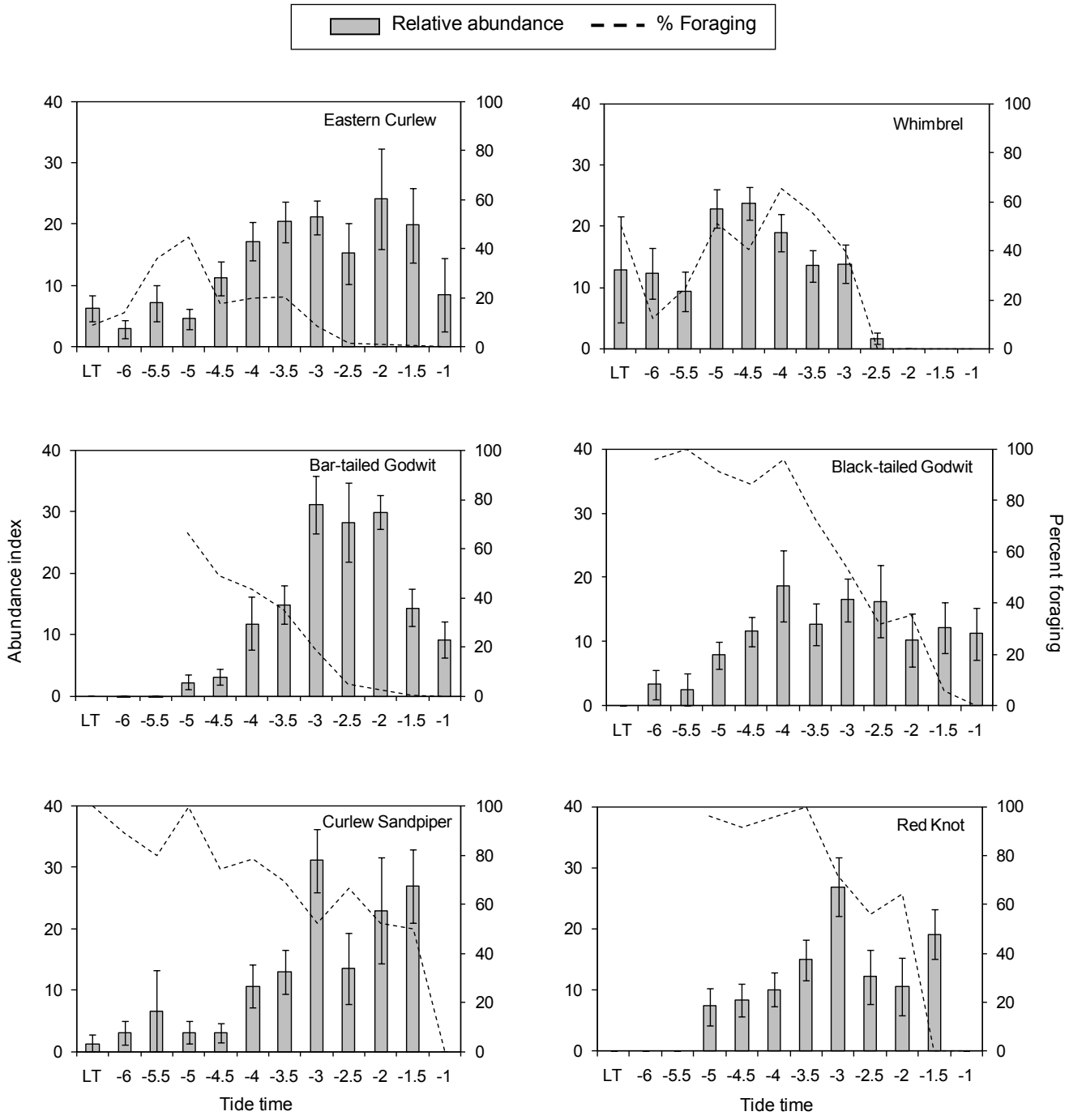


Figure 4.6 Changes in mean abundance (± S.E.) and foraging activity of six migratory shorebird species in the main study area, impoundment two, in relation to tide time. The data are averaged over 20 survey days (Oct-Dec 2004). The abundance index is the number of birds in each survey scan as a proportion of total birds observed in each survey day. Tide time is measured tide height in hours before published high tide (LT = low tide) (see Methods).

Foraging behaviour in the impoundment could partly be explained by body mass. The proportion of foraging birds in flocks was negatively correlated with body mass (r = -0.7, n = 12, p = 0.012), with a higher proportion of feeding activity in small-bodied species compared to larger species (Fig. 4.7).

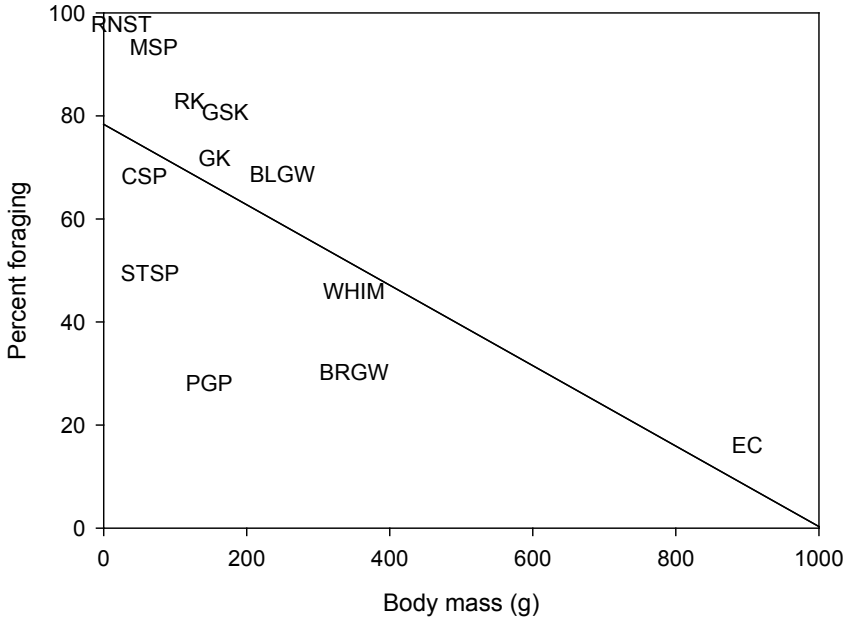


Figure 4.7 Relationship between body mass and flock foraging behaviour in 12 migratory shorebird species in the main mudflat impoundment (line indicates negative correlation). The data are averaged over 20 survey days (Oct-Dec 2004). (Species codes: EC = Eastern Curlew, WHIM = Whimbrel, BRGW = Bar-tailed Godwit, BLGW = Black-tailed Godwit, GSK = Common Greenshank, PGP = Pacific Golden Plover, GK = Great Knot, RK = Red Knot, MSP = Marsh Sandpiper, STSP = Sharp-tailed Sandpiper, CSP = Curlew Sandpiper, RNST = Red-necked Stint).

4.5 Discussion

The importance of supplementary feeding has been well documented for some shorebird species (Dann 1999b; Masero *et al.* 1999; Masero & Perez-Hurtado 2001; Rosa *et al.* 2006). High tide feeding was observed in 12 migratory shorebird species in an artificial mudflat impoundment in the Hunter estuary. Mean bird density was high in the mudflat impoundment (46.6 ± 3.7 birds ha^{-1}) and close to some high tides (18% of flock scans) bird density could exceed 100 birds ha^{-1}). These densities are comparable to high feeding densities (98.3 ± 7.1 birds ha^{-1}) observed on intertidal mudflats in Spain, where some species had to supplement their intake rates by feeding on adjacent supratidal salinas (Masero *et al.* 2000).

In this study, the artificial impoundment supports over 50% of total numbers of migratory shorebirds found in the Hunter estuary in the 2004-05 non-breeding season, including a high proportion (≥ 70%) of Bar-tailed Godwits, Black-tailed Godwits, Red Knots and Curlew Sandpipers. Curlew Sandpipers have declined dramatically (> 80%) in the Hunter estuary and other parts of south-eastern Australia in the last decade (see Chapter 2). This artificial habitat also supported internationally significant numbers (>1% of the EAAF) of Eastern Curlews and nationally significant numbers (>1% of Australian population) of Pacific Golden Plovers (Table 4.2). Large flocks of Red Knots, which pass through the estuary on their way to non-breeding sites further south of the Hunter estuary, and three threatened species: Black-tailed Godwit, Great Knot and Terek Sandpiper (NSW *Threatened Species Conservation Act 1995*) also used the impoundments.

The mudflat impoundments were most important for shorebird species during neap tidal cycles and one to three hours before high tide when the availability of intertidal mudflats was limited in the rest of the Hunter estuary. Some shorebirds switch feeding habitats during neap cycles (Burger *et al.* 1977; Zwarts *et al.* 1990b; Colwell & Mathis 2001), as the tide does not recede as far during neap tides as on spring tides, and so some shorebirds need to feed for longer to meet their energetic requirements before going to roost (Puttick 1980; Evans 1988; Zwarts *et al.* 1990b). Studies tracking Bar-tailed Godwit movements in the Hunter estuary also found that the mudflat impoundments were most important during neap tides and also after storms (Richardson 2004; Foate 2005). In this study, supplementary feeding was most important for small shorebird species, including Red Knots, Curlew Sandpipers and Red-necked Stints (Figs. 4.5; 4.6), which

generally fed as long as mudflats were exposed. Average prey intake rates (mg s^{-1}) generally increase with body mass, with smaller shorebirds taking smaller prey items than larger birds (Zwarts *et al.* 1990a; b). Larger shorebirds generally only need to feed half as long as smaller species (Engelmoer *et al.* 1984; Dann 1987), e.g., six hours in every 24 hour cycle compared to 13 hours for some smaller shorebirds (Zwarts *et al.* 1990b). In addition to taking larger prey, larger species also have lower mass-specific energy requirements and so generally feed for less time (Fig. 4.7).

Foraging efficiency can also be influenced by leg length (maximum 'wadeable depth') (see Table 4.1), experience and perceived predation risk. The mudflat impoundment was not always available for feeding as birds were forced to leave the area when the incoming tide exceeded their preferred wading depth. Generally, shorebird species are uncomfortable in water deeper than their upper thigh and will move to higher grounds (Ntiamoa-Baidu *et al.* 1998). Smaller species departed first, followed by Common Greenshanks, Bar-tailed Godwits and Black-tailed Godwits, while larger Eastern Curlews were often the last species recorded on the mudflat impoundment. Only 16 ± 2.0% of Eastern Curlews foraged in the impoundment and although age ratios were not quantified in this study, these birds were often juveniles. Younger shorebirds generally have lower capture rates and use lower quality habitats than adults (Caldow *et al.* 1999; Stillman *et al.* 2000). As most shorebird species avoid tall vegetation, because of the risk of attack by birds of prey (Dekker 1998; Dekker & Ydenberg 2004), tall fringing vegetation can also limit the area of foraging habitat available to shorebirds. In this study, mudflat in the centre of the impoundment was exposed for longest, which provided extended feeding time for small shorebirds away from the mangrove edge.

The impoundments did not only provide feeding habitat: they also acted as a staging roost for some shorebirds. Large shorebird species are usually forced to rest after a few hours of feeding, to digest their food before they can resume feeding again (Zharikov & Skilleter 2003b; van de Kam *et al.* 2004). Most Eastern Curlews and Bar-tailed Godwits rested and preened in the impoundment (Fig. 4.5) before the incoming tide forced them to move to the main day roosts. These species may prefer to roost on mudflats in the impoundment in preference to the day roosts, where they can suffer high rates of disturbance over the high tide period (see Chapter 3).

The mudflat impoundments supplement other shorebird habitats in the Hunter estuary by providing additional feeding and roosting opportunities, but these artificial habitats are threatened by hydrological changes resulting from major dredging works in the Hunter River and further expansion of coal loading facilities on Kooragang Island. The Kooragang dyke wall, which controls flows into- and out of- the impoundments, is also being eroded by physical and chemical decay processes, which put the estimated lifespan of the dyke wall at 30 years from now (Howe 2008). Without the dyke wall, the mudflat impoundments would function differently and not be available as foraging habitat for shorebirds in the last stages of the low tide period. In light of these major threats, there needs to be greater recognition of the role these artificial mudflats play in maintaining shorebird populations in the Hunter estuary, by ensuring the Kooragang dykes and associated impoundments are protected from development and rehabilitated to maximise habitat availability for migratory shorebirds during their non-breeding seasons.

CHAPTER 5: SHOREBIRD FEEDING DISTRIBUTION AND FORAGING RATES OF BAR-TAILED GODWITS *LIMOSA LAPPONICA* IN THE HUNTER ESTUARY, AUSTRALIA

5.1 Abstract

Many migratory shorebird species spend their non-breeding season (Sept-Apr) along the coastline of Australia. Intertidal mudflats are often their preferred feeding habitat where birds can satisfy their daily energy requirements and build fat reserves for their next migration. The factors that influence feeding habitat selection are complex and can directly affect intake rates. In this study, I investigated migratory shorebird species distribution on six areas of intertidal mudflat in the Hunter estuary, in south-eastern Australia (2005-06). To investigate fine-scale differences between the six study areas, I also measured benthic invertebrate abundance, sediment characteristics and foraging behaviour of a common migratory shorebird, the Bar-tailed Godwit *Limosa lapponica*. There were significant differences in shorebird and benthic invertebrate assemblages, and sediment characteristics in the feeding sites. Bar-tailed Godwits were widely distributed within the Hunter estuary and were associated with at least four other species: Eastern Curlews *Numenius madagascariensis*, Whimbrels *Numenius phaeopus*, Common Greenshanks *Tringa nebularia* and Black-tailed Godwits *Limosa limosa*. Shorebird abundance was lowest at low tide when birds were spread widely over exposed mudflats and highest at staging sites immediately prior to the high tide roosting period. Overall, a high percentage (> 90%) of Bar-tailed Godwits foraged during the low tide period and 50% of birds continued to forage up to three hours after low tide. There were significant differences in the foraging success of Bar-tailed Godwits among the feeding sites which could be explained by the density of their preferred prey or age- and sex-related differences among focal birds. Mean densities of the crab *Macrophthalmus setosus* and the polychaete *Nephtys australiensis* explained some differences in the distribution of some shorebird species; however, further studies of benthic invertebrate assemblages and shorebird prey choice are needed.

5.2 Introduction

Intertidal mudflats provide rich feeding grounds where migratory shorebirds can satisfy their daily energy requirements during their non-breeding season and fuelling requirements prior to migration (Piersma *et al.* 1999; Landys-Ciannelli *et al.* 2003; Battley & Piersma 2005). Many factors affect the selection of feeding habitat (see Fig. 5.1), including prey density and substrate type, which, in turn, can be correlated with shorebird density (e.g, Goss-Custard 1970; Goss-Custard 1977b; Meire & Kuyen 1984; Yates *et al.* 1999; Rogers 2006; Finn *et al.* 2007).

In Australia, most migratory shorebird species forage on intertidal mudflats during low tide periods, day or night (Dann 1987; Rohweder & Baverstock 1996). The availability of their benthic invertebrate prey can be determined by exposure times, state of the tidal cycle, time of day and weather (Dann 1987; Finn 2007). The proximity of feeding grounds to high tide roost sites is important, as shorebirds can minimise their daily energy expenditure by limiting flight distances between their feeding and roosting sites (Lawler 1996; Dias *et al.* 2006; Rogers *et al.* 2006a). Further, the importance of a mudflat can also be related to its area, shape, size and shoreline length, proximity to other high quality feeding habitats and the presence of tall vegetation (i.e., vegetation screens can provide cover for ambushing predators) (Fig. 5.1) (Lawler 1996). The 'quality' of shorebird feeding habitats can be measured by determining the distribution of shorebirds and their prey in relation to substrate types (Thompson 1998; Rogers 1999; Paton *et al.* 2000b; Finn *et al.* 2007) or by quantifying intake rates of individual shorebird species (Zharikov & Skilleter 2002; Masero 2003; Yasue *et al.* 2003; Rohweder & Lewis 2004). Intake rates, however, can vary according to age- or sex-specific differences in shorebird species (Caldow *et al.* 1999; Stillman *et al.* 2000; Zharikov & Skilleter 2002) or daily with bad weather or high disturbance by humans and birds of prey (Burger & Gochfeld 1991; Fitzpatrick & Bouchez 1998; Whitfield 2003a) (Fig. 5.1).

Significant gaps in knowledge remain for many shorebird species that spend their non-breeding season in south-eastern Australia, as there has been a limited number of studies investigating their foraging behaviour (Dann 1987) (for a review see section 1.3). In this chapter, I investigated the distribution of migratory shorebirds in relation to stage of the tidal cycle (tide time) on intertidal mudflats in the Hunter estuary, Australia over two of their non-breeding seasons (2004-2005; 2005-06). Up to 35 migratory shorebird species have been recorded in this non-breeding site

during summer months (Sept-Apr) and the estuary currently supports at least two species in numbers > 1% of their East Asian-Australasian Flyway (EAAF) populations and seven species in nationally significant numbers (> 1% of Australian populations) (see Chapter 2). To characterise the main feeding sites, I measured shorebird and prey abundance, and sediment characteristics in six areas of intertidal mudflat. I also measured foraging rates of the Bar-tailed Godwit *Limosa lapponica baueri*, a common migratory shorebird in the Hunter estuary, to determine the relative importance of each feeding site studied and factors that could influence habitat use and foraging behaviour in this species.

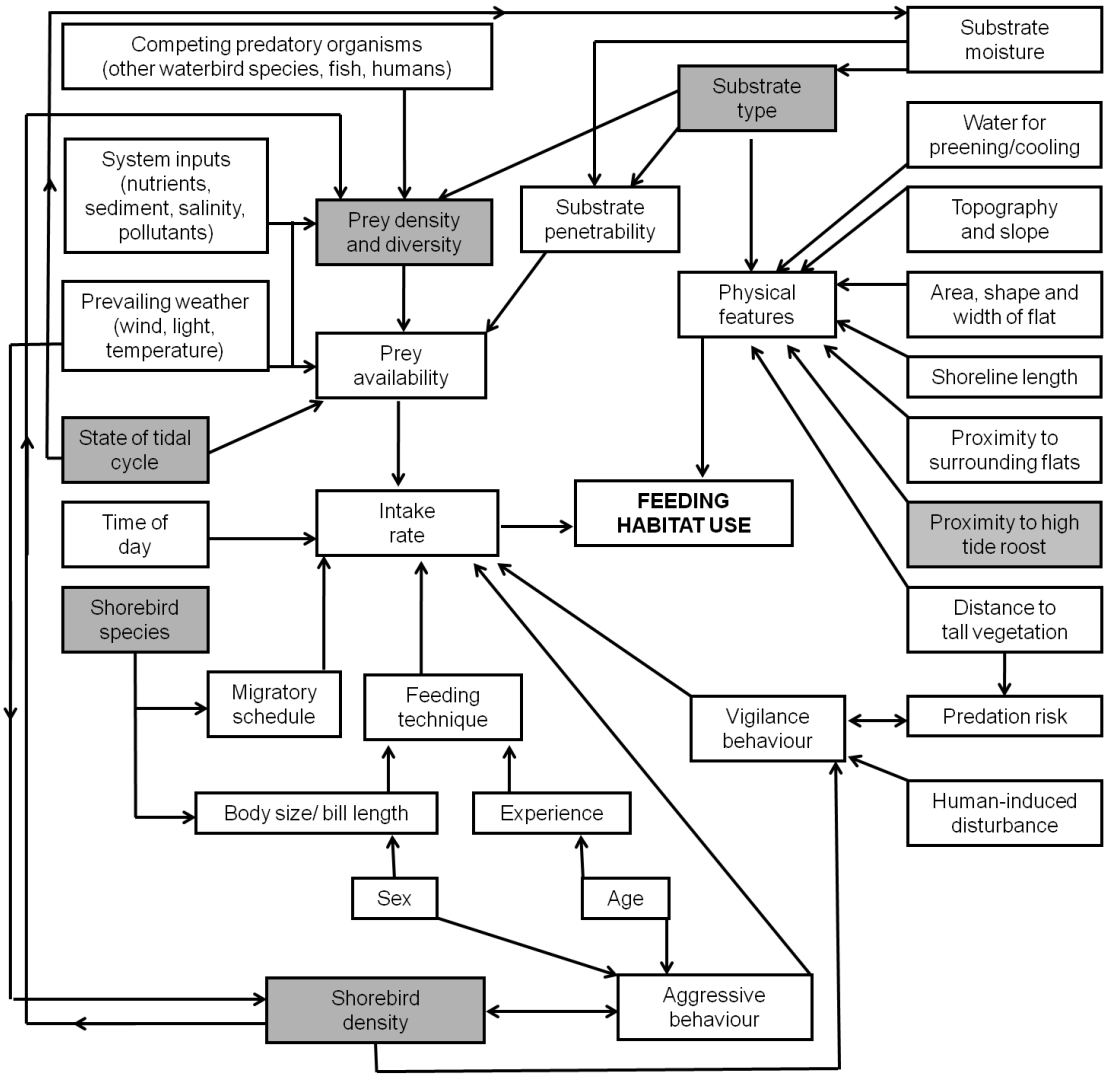


Figure 5.1 Factors influencing intertidal feeding habitat use and intake rates in migratory shorebird species in Australia (shaded boxes are key factors in habitat use) (after Lawler 1996; Finn 2007).

5.3 Methods

5.3.1 Study site

This study was carried out in the Hunter estuary in New South Wales (NSW), south-eastern Australia (32° 51'S/ 151° 46'E) in summer months (Jan-Mar) in 2005 and 2006. The development of the industrial port city of Newcastle has caused significant losses of estuarine habitats in the Hunter estuary, including a 57% reduction in shoreline available to foraging shorebirds (Kingsford & Ferster Levy 1997) (see section 2.2). The Hunter estuary now contains two main river channels that surround Kooragang Island and connect to Fullerton Cove, a large tidal embayment about 9 km north of the mouth of the Hunter River (Fig. 5.2). These estuarine habitats were gazetted as a protected area, the Kooragang Nature Reserve, in 1983, the largest estuarine reserve in NSW at the time (NPWS 1998a). The Kooragang Nature Reserve was recognised as a wetland of international importance under the Ramsar Convention in 1984 and by the East Asian-Australasian shorebird site network in 1996. The Hunter estuary is ranked as the fifth most important site for shorebird species diversity in Australia (Watkins 1993) and the most important site for shorebirds in the NSW (Smith 1991) (see Chapter 2). Both the Kooragang Nature Reserve and adjoining Hexham Swamp were reserved under the Hunter Wetlands National Park (4,255 ha) (Fig. 5.2) in 2007 (DECC 2007).

I recorded migratory shorebird abundance and Bar-tailed Godwit foraging behaviour on six areas of intertidal mudflats in the Hunter estuary. Fullerton Cove receives semi-diurnal tides which expose about 750 ha of intertidal mudflat (Geering 1995) for 5.5 - 6.5 hours twice daily. The tidal range at the mouth of the cove is 1.16 m on spring and 0.82 m on neap tides (Hutchings 1977). A river training wall, the Kooragang dykes (Fig. 5.2), impounds four ponds which hold an additional 25 ha of intertidal mudflats. During day-time high tides, most shorebirds roost on the Kooragang dykes or Stockton sandspit (Fig. 5.2) in the main river channel (see Chapter 3). Large sections of the rock wall have eroded since its construction and breaks in the wall allow the ponds to fill and drain with the tide, exposing mudflats for longer periods than other mudflats in the rest of the Hunter estuary and extending foraging time for some shorebird species (see Chapter 4). I observed four feeding sites within Fullerton Cove and two sites impounded by the river training wall (Fig. 5.2). Each mudflat area was chosen based on it being a representative distance from the main day roosts in the North Arm of the Hunter River, with two feeding sites about 6.5 km away (sites 2 and 6), two sites about 4 km away (sites 1 and 5) and two sites (sites 3 and 4) were

close (<1 km) to the main day roosts (see Figure 5.2). Each feeding site was a stretch of intertidal mudflat about 500 m in length and fringed by large mangroves.

5.3.2 Shorebird observations

I surveyed feeding sites 1 - 3 in the 2005 season (Jan – Mar 2005; N = 14) and sites 1 - 6 in the 2006 season (Jan – Mar 2006; N = 21) for migratory shorebirds (200 observation hours, 771 flock scans in total). Sampling days were randomly allocated to each feeding site. Tide heights ranged from 0.13 - 0.70 m (BOM 2005) and sampling was limited to daylight hours. Feeding sites in Fullerton Cove (FC) (sites: FC1, FC2, FC5 and FC6) were observed from a four metre inflatable boat about 400 m from the mangrove edge using binoculars (8 x 30 mm) and a telescope (Swarovski 20 – 60X zoom). I observed shorebirds in the largest mudflat impoundments behind the Kooragang dykes (KD) (sites KD3 (impoundment two) and KD4 (impoundment three) from the rock wall using binoculars and a telescope. Observations of the Fullerton Cove sites started three hours before and finished about three hours after published low tide (at Newcastle). Adverse weather conditions prevented completion of four surveys in Fullerton Cove. Due to the lag in the tidal regime in the mudflat impoundments, I observed sites KD3 and KD4 from one hour before and up to six hours after published low tide.

Flock scans were at 15 minute intervals at the feeding sites (Altmann 1974), with randomisation of direction. During each scan, total counts and number of foraging birds were recorded for each shorebird species. Foraging birds were pecking/ probing the substrate, handling a prey item or visually searching for prey. Markers fixed to mangroves at the Fullerton Cove sites marked the boundaries of each site while dividing walls between the mudflat impoundments were boundaries for sites KD3 and KD4 (Fig. 5.2). A PVC pipe (2.4 m x 90 mm) driven vertically into the mud at each feeding site, allowed measurement of changes in relative tide height and time of tide reversal (measured in hours before (- values) and after (+ values) low tide (low tide = 0)) (Burger *et al.* 1977). The presence of birds of prey was also noted during each flock scan.

I used focal animal sampling of Bar-tailed Godwits to investigate site effects, microhabitat associations, and age and sex effects. Total numbers of Bar-tailed Godwits reach about 1,400 during summer months (Sept-Apr) (see Chapter 2). Actively foraging Bar-tailed Godwits were selected randomly, observed for one minute (Altmann 1974) in 2005 (n = 99) and 2006 (n = 263) and observations dictated onto a micro-cassette recorder and transcribed afterwards. Foraging

was defined as all pecks and probes in and on the sediment whilst captures related to any visible prey items or swallows seen in foraging birds. Microhabitat and nearest-neighbour distance (relative to bird length) and species were also recorded at the start of each focal sample. Microhabitat included: mudflat; mangrove/mudflat (within 10 m of mangrove fringe); mudflat/open water (tide edge); or open water. When birds stopped feeding, left the area or were disturbed, focal samples were discarded. Ages (adult/juvenile) and sex were determined for focal birds in the 2006 season only (n = 195). Ageing was based on the presence of breeding plumage, size and the degree of pre-migratory fattening, while sex determination was from overall size and bill length, which are greater in female compared to male Bar-tailed Godwits (Barter 1989b, a; Higgins & Davies 1996).

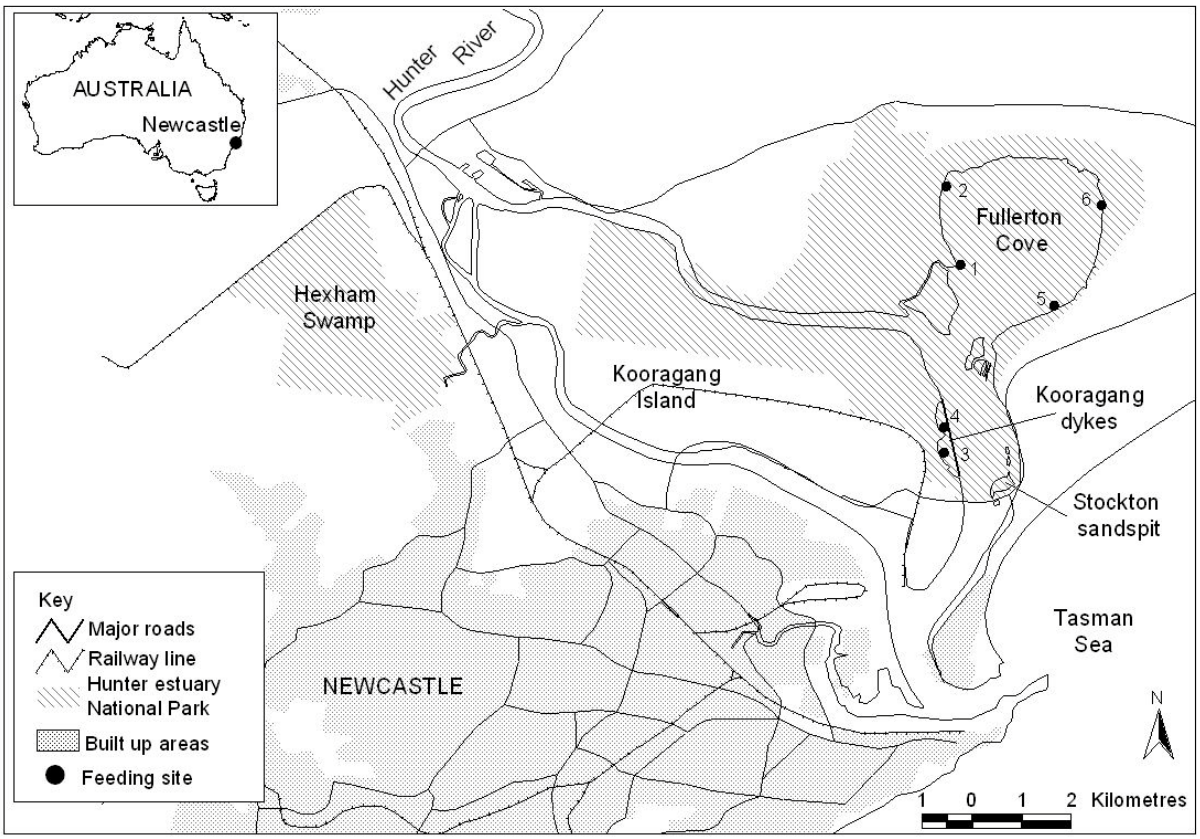


Figure 5.2 Location of the six intertidal mudflat study sites and major day roosts (Kooragang dykes and Stockton sandspit), in the Hunter estuary, New South Wales, Australia.

5.3.3 Benthic invertebrates

I collected benthic invertebrates from the six feeding sites in January 2006. Three plots about 50 m apart along the water's edge, where most shorebird species concentrate, were sampled in each feeding site. A PVC corer (90 mm diameter) was driven into the mud to a depth of 30 cm to collect six cores at each plot, haphazardly in a 5 m radius around each plot. This depth and sample size obtained the highest number of species per sampling effort in a pilot study in December 2005 (see Appendix D). A thick layer of oysters prevented samples deeper than 5 cm being collected from site FC5 (Fig. 5.2). Cores were sealed in plastic bags and sorted on the day of collection using 0.5 mm open-ended sieve bags (430 x 270 mm) (Sefar Filter Specialists, Blacktown, Australia). All invertebrates were picked live from sorting trays, preserved in 70% ethanol and identified to the lowest taxonomic group possible (P. Hutchings and R. Springthorpe, Australian Museum, Sydney assisted with species identifications and I used Robinson and Gibbs (1982) to identify molluscs). Numbers of invertebrates were expressed as individuals per m^2 this was based on a total sampled surface of 0.1 m^2 per feeding site. The total number of crab holes (>3 cm diameter) and percentage algal mat cover was also estimated for each plot with five 1 m^2 quadrats positioned haphazardly along the water's edge. Previous studies have shown that burrow counts are a good surrogate for estimating total crab abundance in intertidal habitats (Warren 1990; MacFarlane 2002; Mazumder & Saintilan 2003).

5.3.4 Sediment sampling

An extra core (30 cm depth, 55 mm PVC pipes) was taken from each plot of each feeding site, sealed and frozen for laboratory analysis. After defrosting and air-drying to a constant mass, samples were processed to determine the percentage coarse (>2.36 mm) and silt fractions (< 0.068 mm), median particle size, percentage total organic carbon, conductivity and pH. For analysis of particle size distribution, a sub-sample of air-dried sediment was crushed using a porcelain mortar and pestle for ten minutes, placed in the top of a stainless steel stacked sieve set (2.36, 1.18, 0.600, 0.300, 0.150 and 0.075 mm mesh sizes) and shaken for ten minutes using a mechanical shaker. These sieve fractions were then oven-dried at 40 ^oC to a constant mass to determine the proportion of each fraction. Two separate air-dried sub-samples from the 0.300 mm sieve were analysed using a Malvern Mastersizer E volumetric laser diffraction particle analyser to determine the particle size distribution of this fine fraction. The percentage of organic matter in each sample was estimated by loss of mass on ignition. Two 5 g replicates of each

sample were placed in an electric muffle furnace and heated to 550 ^{o}C for two hours. The mass of each replicate was recorded after ignition and the percentage total organic carbon calculated as a percentage of the pre-ignition mass (Smith & Atkinson 1975). Sediment electrical conductivity and pH were measured in a 1:5 soil to water solution by weight (Hazelton & Murphy 1992). Two 5 g replicate air-dry soil samples were placed in specimen tubes with 25 ml of de-ionised water. The tubes were sealed and shaken vigorously at 10 minute intervals for one hour. After the soil slurry had settled, the pH of the solution was measured using an Orion 720 pH meter and conductivity measured in the second sample using an Orion 1230 EC probe.

5.3.5 Statistical analyses

Differences in shorebird and invertebrate species assemblages at the feeding sites were examined using the Bray-Curtis measure of similarity (Bray & Curtis 1957). Shorebird flock scans were averaged for each survey (shorebird counts were pooled from the two survey years (2005 and 2006) to provide sufficient replication for each feeding site). Benthic invertebrate data represented average counts of each species per plot. Both sets of abundance data were transformed with fourth root to control for multiple zeros and large values present in the data sets (Quinn & Keough 2002). The transformed abundance data were subjected to non-metric multi-dimensional scaling (nMDS) to demonstrate patterns in shorebird and benthic invertebrate species assemblages at the feeding sites (Clarke & Warwick 2001). One-way analysis of similarity tests (ANOSIM) (Clarke & Warwick 2001) were used to detect significant differences in species assemblages among sites. The contribution made by particular species to identified differences at the sites was determined by analysis of similarity percentages (SIMPER). Euclidean distances matrices were used to construct nMDS plots for abiotic variables (conductivity, pH, total organic carbon, mean particle size and % silt) following square root transformations. The relationships between benthic invertebrate assemblages and abiotic variables were investigated further with the biological-environmental linkage (BIOENV) procedure (Clarke & Warwick 2001). This procedure selects the best subsets of abiotic variables that maximise Spearman rank correlations coefficients (r_s) between the similarity matrix of the transformed invertebrate abundances per plot and all possible dissimilarity matrices of the abiotic data assembled using Euclidean distances (Clarke & Ainsworth 1993). After transformation of the data (log10 (x+1)), I used two-tailed Pearson's correlations (Quinn & Keough 2002) to investigate whether shorebird abundance was correlated with the five most abundant invertebrate species (*Victoriopisa australiensis, Arthritica*

helmsi, *Scolopos simplex*, *Nephtys australiensis* and *Macrophthalmus setosus*) in the feeding sites.

Due to non-normal distributions, I used non-parametric Kruskal-Wallis tests, comparing the *H*-statistic to chi-square distribution (Quinn & Keough 2002), to investigate differences in flock size and the proportion of foraging godwits among survey sites. Two measures of foraging behaviour were calculated from observations of focal Bar-tailed Godwits; (i) foraging rate, the sum of all bill movements (pecks and probes) per minute; and (ii) the total number of captures per minute. I was not able to build a robust general linear model to determine the significance of each factor (i.e., site, microhabitat, age and sex and their interactions) on godwit foraging behaviour, as there was an unequal number of focal samples among the treatments (Underwood 1997). Instead, I used separate Kruskal Wallis tests for non-normal data and one-way Analysis of Variance (ANOVA) tests for transformed data (log10(x+1)) meeting the assumptions of normality and homogeneous variances (Fowler *et al.* 1998). I also investigated whether nearest-neighbour distances affected foraging behaviour of focal birds with a two-tailed Pearson's correlation (Quinn & Keough 2002). Univariate analysis were done in SPSS (2005) and multivariate analyses in PRIMER (2002). Means and standard errors (S.E.) are presented throughout.

5.4 Results

5.4.1 Shorebird distribution

There were significant differences in shorebird assemblages at the feeding sites (*global R* = 0.563, $p = 0.001$) (Fig. 5.3a). Twelve migratory shorebird species were recorded on the intertidal mudflats. Bar-tailed Godwits were the most abundant and widespread species observed during surveys. Mean counts of Bar-tailed Godwits were highest at sites FC1 and FC2 in Fullerton Cove, but large flocks (maximum count 685 birds) were also recorded at site KD3 in the late stages of the low tide period. Bar-tailed Godwits were often closely associated with Eastern Curlews *Numenius madagascariensis*, Whimbrels *Numenius phaeopus*, Common Greenshanks *Tringa nebularia* and Black-tailed Godwits *Limosa limosa*. These species accounted for more than 90% of similarity among species assemblages at the feeding sites (Fig. 5.3b; Table 5.1). Red-necked Stints *Calidris ruficollis* and Marsh Sandpipers *Tringa stagnatilis* were disassociated from the other species (Fig. 5.3b) as they were only recorded at site KD3 (Table 5.2).

Most shorebird species were recorded on site KD3 (12 species) at the Kooragang dyke ponds. In the Fullerton Cove sites, the highest number of species (nine species) was recorded at the western entrance of the cove at site FC1. Sites FC5 and FC6 on the eastern side of Fullerton Cove had the lowest species diversity (four species) (Table 5.2). Sites KD3 and KD4 at the Kooragang dyke ponds were important for small and medium-sized shorebirds, including Curlew Sandpipers *Calidris ferruginea*, Pacific Golden Plovers *Pluvialis fulva* and Sharp-tailed Sandpipers *Calidris acuminata*, throughout the low tide cycle. Some shorebird species were only recorded at one site in Fullerton Cove (FC1) and maximum counts of these species were generally low: Red Knot *Calidris canutus* (59 birds); Pacific Golden Plover (15 birds); Curlew Sandpiper (4 birds); and Great Knot *Calidris tenuirostris* (3 birds) (Table 5.2).

Shorebird counts were lowest at low tide when most shorebirds were spread out around the estuary and highest at staging sites either side of the low tide period when mudflat availability was most limited (Fig. 5.4). Sites FC1, KD3 and KD4 were used as staging sites by many shorebirds outside of the core low tide period (Fig. 5.4). Site FC1 had bimodal peaks in total numbers of migratory shorebirds reflecting birds moving into and out of Fullerton Cove. Sites KD3 and KD4 were most important in the later stages of the low tide period when sites in Fullerton Cove were unavailable (Fig. 5.4). All mudflats in Fullerton Cove were covered by

about 3.75 hours after low tide, but mudflats in sites KD3 and KD4 were available for longer (Fig. 5.4). At low tide, total counts of migratory shorebirds were very low or absent in the Kooragang dyke ponds (sites KD3 and KD4), but mudflat area was maximised here at two hours after published low tide (Fig. 5.4). From around 2.5 hours after low tide, large flocks of Eastern Curlew and Bar-tailed Godwit arrived at sites KD3 and KD4 from Fullerton Cove. Migratory shorebirds used sites FC5 and FC6 on the eastern side of Fullerton Cove later in the low tide cycle (one hour before low tide) compared to sites FC1 and FC2 on the western shore where birds were observed foraging two hours before low tide (Fig. 5.4).

Table 5.1 SIMPER (Bray-Curtis similarity percentages) analysis of mean shorebird abundance among the six feeding sites (FC = Fullerton Cove; KD = Kooragang Dykes).

Site	Common name	Mean abundance	Contribution %	Cumulative %
FC1	Bar-tailed Godwit	84.9	60.5	60.5
	Eastern Curlew	23.8	27.0	87.7
	Black-tailed Godwit	7.6	6.2	93.7
FC2	Bar-tailed Godwit	36.4	84.3	84.3
	Eastern Curlew	2.1	6.3	90.6
KD3	Bar-tailed Godwit	91.1	46.0	46.0
	Eastern Curlew	36.2	25.0	71.0
	Curlew Sandpiper	34.2	12.9	83.9
	Black-tailed Godwit	73.4	11.8	95.7
KD4	Bar-tailed Godwit	26.4	76.8	76.8
	Common Greenshank	3.4	11.1	87.9
	Eastern Curlew	2.8	7.8	95.8
FC5	Bar-tailed Godwit	1.8	50.2	50.2
	Common Greenshank	1.0	33.1	83.3
	Whimbrel	0.4	14.8	98.0
FC6	Bar-tailed Godwit	8.7	59.5	59.5
	Eastern Curlew	1.1	24.2	83.6
	Common Greenshank	1.0	12.2	95.8

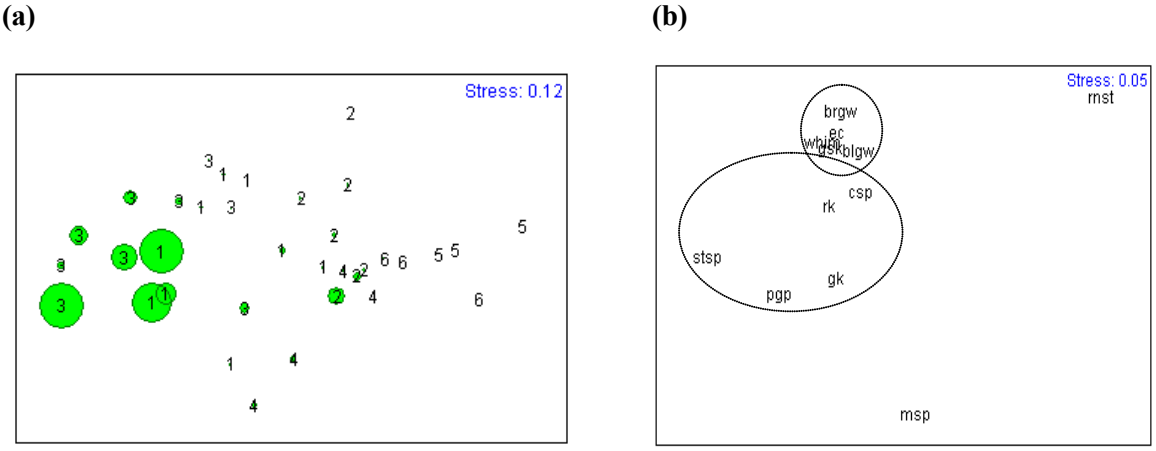


Figure 5.3 Non-metric multi-dimensional scaling plots showing: (a) patterns in shorebird assemblages in the feeding sites (1- 6) (shaded circles represent mean counts of Bar-tailed Godwits per survey period); and (b) species associations across all feeding sites (see Table 5.2 for species codes). Data are combined from the 2005 and 2006 non-breeding seasons.

Table 5.2 Mean (± S.E.) and maximum counts of 12 shorebird species recorded at six intertidal mudflat feeding sites (FC = Fullerton Cove, KD = Kooragang Dykes) compared to total population estimates for the Hunter estuary (HE) (Jan-Mar in 2005 and 2006). Mean counts are based on low tide counts only (maximum counts are from all counts in each six hour observation period) (n = number of surveys).

Common name (species codes)	FC1 (n = 9)			FC2 (n = 7)			KD3 (n = 9)			KD4 (n = 4)			FC5 (n = 3)			FC6 (n = 3)			HE popn.*
	mean	(SE)	max	mean	(SE)	max	mean	(SE)	max	mean	(SE)	max	mean	(SE)	max	mean	(SE)	max	
Bar-tailed Godwit (BRGW)	54.4	(22.3)	613	47.9	(38.3)	455	0	-	685	2.0	(2.0)	209	14.3	(5.5)	25	9.3	(7.0)	57	1103
Black-tailed Godwit (BLGW)	22.7	(22.4)	202	0	-	154	44.1	(27.8)	241	0	-		0	-		0	-		225
Curlew Sandpiper (CSP)	0.4	(0.4)	4	0	-		2.6	(1.8)	94	8.5	(8.5)	200	0	-		0	-		381
Eastern Curlew (EC)	15.4	(6.3)	205	1.5	(0.7)	8	2.3	(1.4)	333	2.3	(1.4)	9	1.3	(0.9)	3	1.7	(0.7)	14	391
Great Knot (GK)	0.1	(0.1)	3	0	-		0	-	3	0	-		0	-		-	-		4
Common Greenshank (GSK)	0.3	(0.2)	40	1.4	(0.4)	15	11.1	(4.7)	46	3.3	(2.0)	18	1.3	(0.7)	14	1.7	(0.3)	17	166
Marsh Sandpiper (MSP)	0	-		0	-		2.1	(2.1)	31	0	-		0	-		0	-		160
Pacific Golden Plover (PGP)	0	-	15	0	-		0	-	166	39.7	(34.4)	132	0	-		0	-		239
Red Knot (RK)	6.1	(3.8)	59	0	-		0.2	(0.1)	81	0	-	6	0	-		0	-		70
Red-necked Stint (RNST)	0	-		0	-		0	-	1	0	-		0	-		0	-		74
Sharp-tailed Sandpiper (STSP)	0	-		0	-		0	-	14	0	-	19	0	-		0	-		665
Whimbrel (WHIM)	3.2	(0.3)	8	0.6	(2.4)	15	1.3	(0.4)	5	0.3	(0.3)	4	1.3	(0.3)	2	0.3	(0.3)	1	105
Total species	9			5			12			8			4			4			19

*Maximum counts recorded during monthly summer counts (Jan-Mar) in 2005 and 2006 (n = 6) by the Hunter Bird Observers Club (Stuart 2006, 2007).

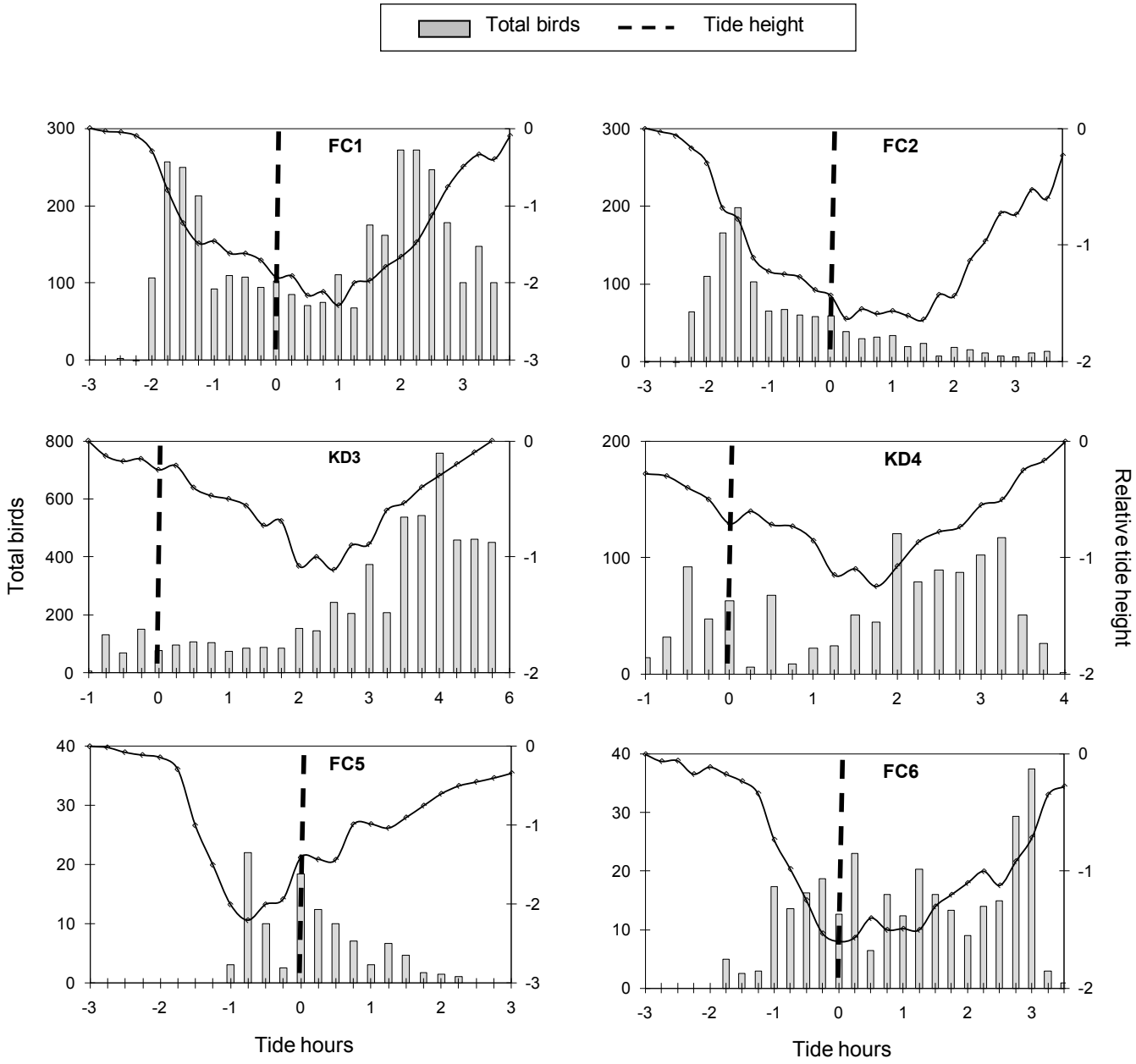


Figure 5.4 Temporal changes in the mean total number of migratory shorebirds and in the mean relative tide height at each mudflat feeding site in the Hunter estuary (Jan-Mar 2005; 2006). Tide time is measured tide height in hours before (- values) and after (+ values) published low tide (= 0) (low tide is marked by vertical dashed line) (see Methods).

5.4.2 Site characteristics

Benthic invertebrate diversity was generally low (5 - 12 species), but there were significant differences in benthic invertebrate assemblages among the study sites (g*lobal* $R = 0.701$, $p = 0.001$) (Fig. 5.5). Overall, average dissimilarities between group comparisons were low (22.2 - 42.5%) and the two-dimensional nMDS plots gave a poor representation of the site relationship in the biotic data (stress 0.16) (Fig. 5.6a,b). Although the abiotic variables did not fully explain the patterns in benthic invertebrate assemblages, the multivariate analysis did indicate that of all the abiotic variables sampled, pH and conductivity correlated best with benthic invertebrate species distribution (BIOENV $r_s = 0.50$) (Fig. 5.6c).

Four invertebrate species (*Victoriopisa australiensis, Arthritica helmsi*, *Scolopos simplex*, *Nephtys australiensis*) accounted for more than 90% of similarity among species assemblages at the feeding sites (Table 5.3). Invertebrate diversity was highest at sites FC1 (11 species) and KD3 (12 species) but invertebrate abundance was highest at sites KD4 and FC2 (Fig. 5.5; Table 5.4). Polychaete worms, in particular the Orbiniidae *Scolopos simplex*, were found in large numbers (2829.3 ± 396.0 per m^2) at site KD4. Small bivalves *Arthritica helmsi* (1013.0 ± 143.9 per m^2) and amphipods *Victoriopisa australiensis* (1772.7 ± 221.5 per m^2) were abundant at site FC2 in Fullerton Cove. Sites FC5 and FC6 had the lowest species diversity, seven and five species respectively (Table 5.4). Crab hole density was highest at site FC1 (10.6 ± 1.4 per m^2) where the silt fraction was lowest (37.9 ± 3.3%) (Table 5.4; Fig. 5.6e).

The distribution of algal mats was patchy, but algal growth was present in quadrats at sites FC2 and KD3. Excluding site FC1, the silt fraction was very high (>76%) at the feeding sites. A large coarse fraction was recorded at site FC5 only (8.5 ± 4.1%) where a thick layer of oyster shells was present in the substrate (Table 5.4). The Kooragang dyke wall limited connectivity between sites KD3 and KD4 and the main river channel, which may have explained their higher conductivity and total organic carbon readings (Table 5.4; Fig. 5.6c). Bird of prey sightings were generally low across all survey sites (20 - 30% of flock scans) but included: White-bellied Sea-eagles *Haliaeetus leucogaster*; Peregrine Falcons *Falco peregrinus*; Whistling Kites *Haliastur sphenurus*; Wedge-tailed Eagles *Aquila audax*; and Swamp Harriers *Circus approximans*.

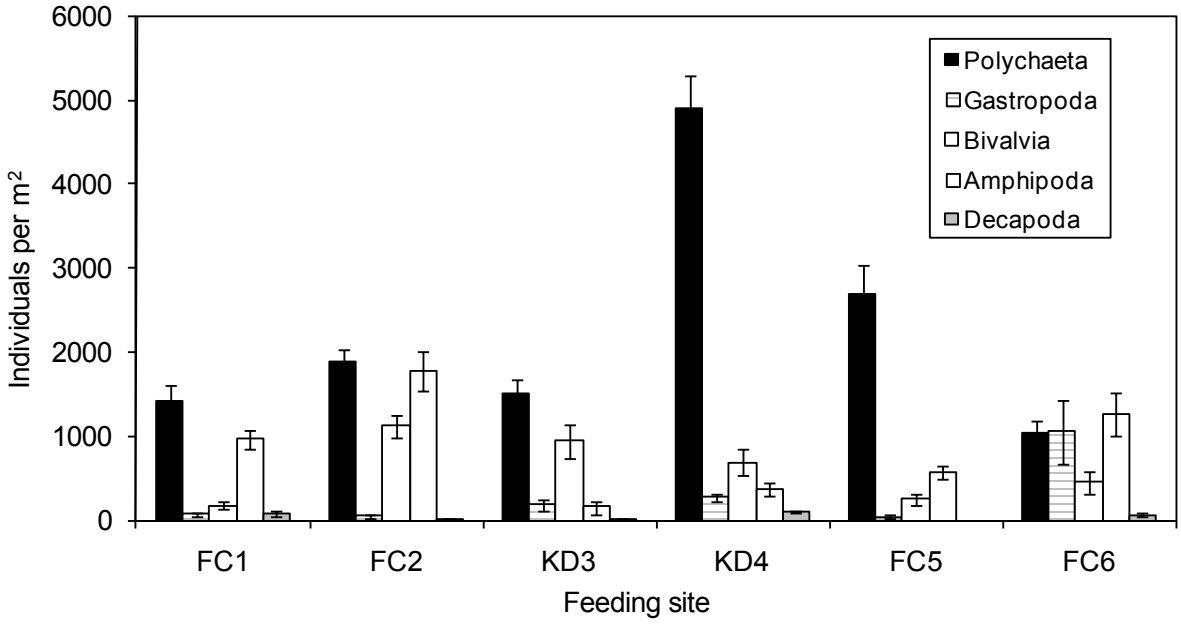


Figure 5.5 Mean (± S.E.) density (per m^2) of each benthic invertebrate group collected from the six feeding sites in the Hunter estuary (January 2006).

Table 5.3 SIMPER (Bray-Curtis similarity percentages) analysis of mean benthic invertebrate abundance (per site) among the six feeding sites (FC = Fullerton Cove; KD = Kooragang Dykes).

Site	Species	Mean abundance	Contribution %	Cumulative %
FC1	*Victoriensis australiensis*	6.2	49.3	49.3
	Nephtys australiensis	3.5	32.6	81.9
	Scoloplus simplex	1.0	9.5	91.4
FC2	*Victoriensis australiensis*	11.3	49.4	49.4
	Arthritica helmsi	6.4	28.6	78.0
	Nephtys australiensis	2.8	11.7	89.6
	Scoloplus simplex	2.3	9.8	99.4
KD3	*Nephtys australiensis*	2.9	39.1	39.1
	Arthritica helmsi	5.2	31.2	70.3
	Scoloplus simplex	2.0	19.2	89.5
	Victoriensis australiensis	1.0	4.3	93.8
KD4	*Scoloplus simplex*	18.3	63.3	63.3
	Arthritica helmsi	7.5	19.1	82.4
	Nephtys australiensis	3.2	9.3	91.7
FC5	*Scoloplus simplex*	8.4	58.8	58.8
	Victoriensis australiensis	4.6	25.6	84.4
	Arthritica helmsi	1.6	9.9	94.3
FC6	*Victoriensis australiensis*	7.7	67.3	67.3
	Scoloplus simplex	2.2	19.4	86.7
	Nephtys australiensis	0.6	5.6	92.3

Table 5.4 Mean (± S.E.) (a) benthic invertebrate density (per m^2); and (b) sediment characteristics of feeding sites in the Hunter estuary (Jan 2006).

(a) Benthic invertebrates		Feeding site[a]					
		FC1	FC2	KD3	KD4	FC5	FC6
Polychaeta	Nereididae: *Simpliseta aequitsetis*	0 (-)	0 (-)	8.7 (8.7)	9.2 (9.2)	69.9 (26.1)	0 (-)
	Neanthes glandiancta	8.7 (8.7)	0 (-)	0 (-)	0 (-)	0 (-)	0 (-)
	Nephtyidae: *Nephtys australiensis*	550.1 (61.3)	436.6 (53.2)	462.8 (50.0)	517.8 (96.0)	139.7 (40.0)	96.1 (22.5)
	Orbiniidae: *Scolopos simplex*	165.9 (34.7)	366.8 (47.5)	305.6 (60.2)	2829.3 (396.0)	1327.3 (151.0)	340.6 (46.3)
	Capitellidae: *Barantolla lepte*	17.5 (12.0)	0 (-)	8.7 (8.7)	0 (-)	0 (-)	0 (-)
Gastropoda	Buccinidae: *Nassarius burchardi*	8.7 (8.7)	0 (-)	8.7 (8.7)	83.2 (27.4)	8.7 (8.7)	0 (-)
	Amphibolidae: *Salinator fragilis*	0 (-)	0 (-)	113.5 (64.6)	27.7 (15.0)	0 (-)	0 (-)
Bivalvia	Leptonidae: *Arthritica helmsi*	96.1 (25.9)	1013.0 (143.9)	820.8 (195.4)	1155.8 (264.8)	253.2 (55.6)	87.3 (36.4)
	Tellinidae: *Tellina deltoidalis*	26.2 (14.2)	17.5 (12.0)	52.4 (18.0)	0 (-)	0 (-)	0 (-)
	Corbulidae: *Corbula vicaria*	26.2 (14.2)	17.5 (12.0)	8.7 (8.7)	0 (-)	0 (-)	0 (-)
	Psammobiidae: *Sanguinolaria donacioides*	8.7 (8.7)	8.7 (8.7)	0 (-)	0 (-)	0 (-)	0 (-)
Amphipoda	Gammaridae: *Victoriopisa australiensis*	969.3 (108.0)	1772.7 (221.5)	148.5 (80.8)	379.1 (80.9)	724.8 (98.5)	1213.8 (268.6)
Decapoda	Callianassidae: *Trypaea australiensis*	0 (-)	0 (-)	17.5 (12.0)	0 (-)	0 (-)	0 (-)
	Ocypodidae: *Macrophthalmus setosus*	69.9 (29.0)	26.2 (14.2)	17.5 (12.0)	37.0 (16.7)	0 (-)	69.9 (26.1)
Total species		11	8	12	8	6	5

(b) Sediment characteristics		Feeding sites[a]					
		FC1	FC2	KD3	KD4	FC5	FC6
	Crab burrow density[b]	10.6 (1.4)	1.0 (0.4)	1.1 (0.3)	6.6 (0.8)	1.7 (0.7)	1.1 (0.4)
	Algal mat cover (%)	0 (-)	87.0 (1.4)	90.3 (1.1)	0 (-)	0 (-)	0 (-)
	Coarse fraction (%)[c]	0.3 (0.1)	0.4 (0.1)	0.5 (0.2)	0.1 (0.1)	8.5 (4.1)	0.2 (0.1)
	Silt fraction (%)[d]	37.9 (3.3)	79.6 (2.2)	92.5 (1.1)	83.4 (1.0)	76.7 (2.2)	83.8 (1.7)
	Median particle size (μm)	104.0 (14.1)	23.0 (1.6)	13.9 (1.2)	19.8 (0.9)	22.4 (3.2)	19.9 (2.3)
	Total organic carbon (%)	4.0 (0.3)	6.6 (0.3)	11.0 (0.2)	8.6 (0.2)	3.5 (0.1)	6.7 (0.1)
	Conductivity (mS/cm)[e]	4.2 (0.2)	5.8 (0.3)	9.0 (0.1)	7.5 (0.1)	3.7 (0.1)	5.8 (0.1)
	pH	7.3 (0.1)	8.0 (0.1)	6.2 (0.2)	7.4 (0.1)	8.5 (0.0)	8.3 (0.0)

[a] Feeding sites: FC = Fullerton Cove, KD = Kooragang dyke ponds. [b] mean (± S.E.) number of crab burrows per m^2; [c] mean (± S.E.) mass percentage of the coarse fraction is the sieved fraction > 2.36 mm; [d] mean (± S.E.) mass percentage of silt fraction is based on fraction < 68μm; [e] electrical conductivity of sediment solution (see Methods).

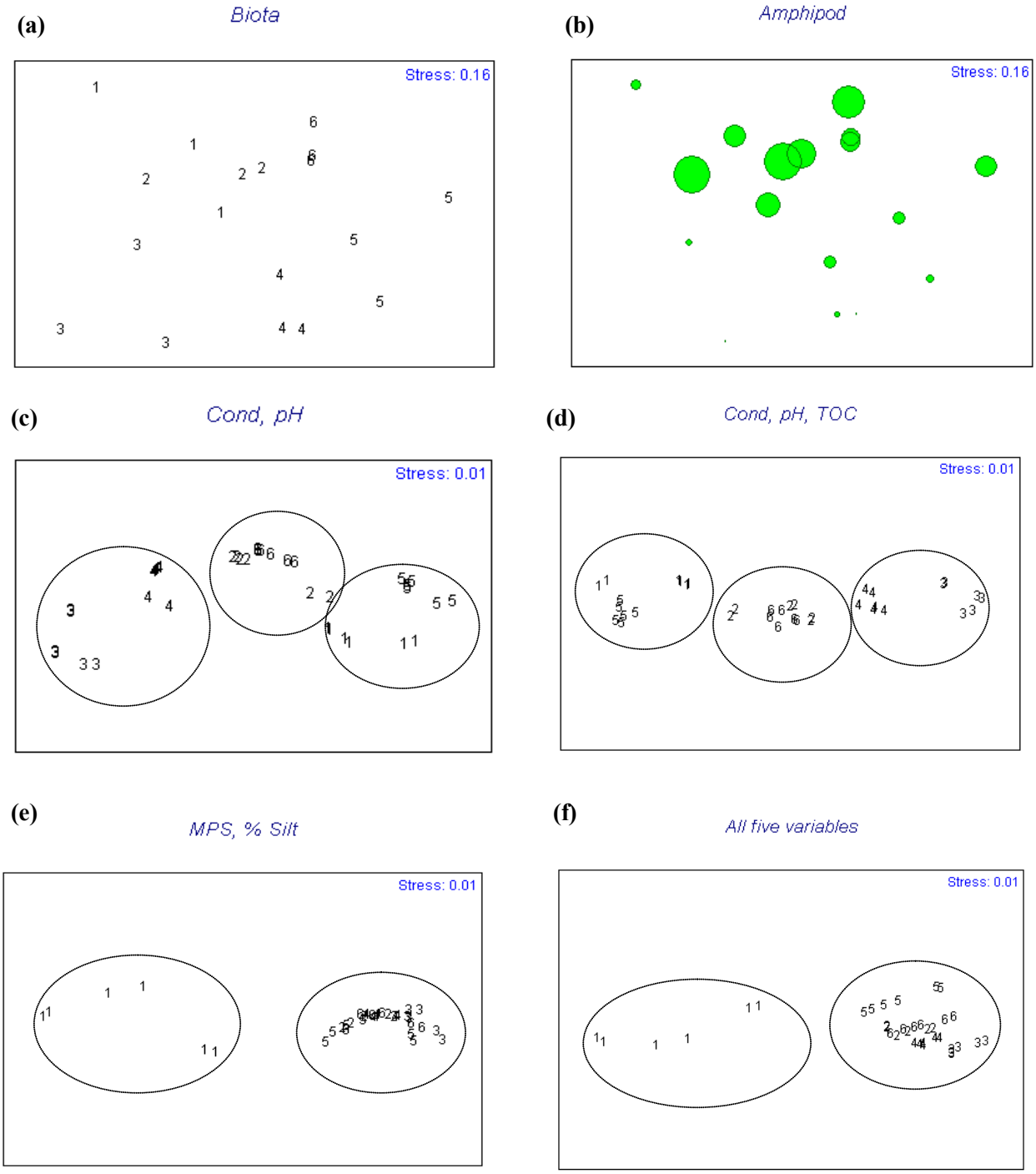


Figure 5.6 Non-metric multi-dimensional scaling plots of the feeding sites (sites 1 - 6) based on the mean (a) abundance of 14 benthic invertebrate species (based on average counts of each species per plot), and (b) the amphipod *Victoriopisa australiensis*, and abiotic variables (based on six samples per site): (c) conductivity and pH; (d) conductivity, pH and total organic carbon (TOC); (e) median particle size (MPS) and % silt content; and (f) all five abiotic variables.

Analysis of relationships between the density of the five most abundant benthic invertebrates and shorebird abundance in the feeding sites revealed some positive correlations. There were significant correlations between the density of the polychaete *Nephtys australiensis* and the abundance of Bar-tailed Godwits (r = 0.8, n = 6, p = 0.033), Whimbrels (r = 0.9, n = 6, p = 0.024) and Red Knots (r = 0.9, n = 6, p = 0.021) in the feeding sites (Fig. 5.7a). There were also significant correlations between the density of the crab *Macrophthalmus setosus* and the abundance of Bar-tailed Godwits (r = 0.9, n = 6, p = 0.020) and Whimbrels (r = 0.9, n = 6, p = 0.017) (Fig. 5.7b). However, there were no significant relationships between the abundance of any of the 12 shorebird species and the density of the polychaete *Scolopos simplex*, the bivalve *Arthritica helmsi* or the amphipod *Victoriensis australiensis*.

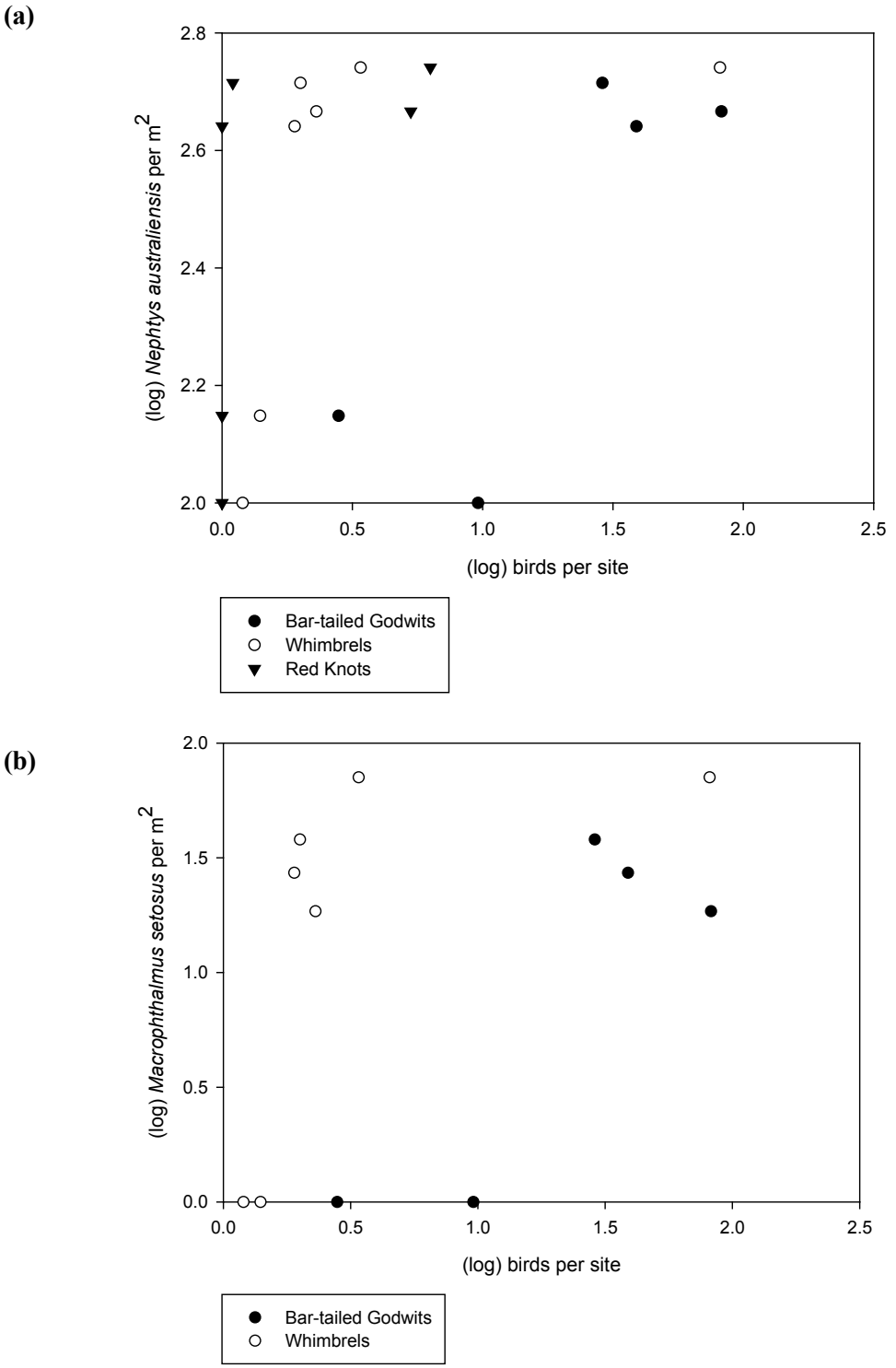


Figure 5.7 Relationships between the mean abundance of Bar-tailed Godwits, Whimbrels and Red Knots and the mean density of the (a) polychaete *Nephtys australiensis* and (b) crab *Macrophthalmus setosus*.

5.4.3 Bar-tailed Godwit foraging behaviour

Bar-tailed Godwits foraged in loosely spaced flocks (8.2 ± 0.4 bird lengths; n = 361; range 0.5 – 40 birds) (Table 5.5) but there were significant differences in mean flock sizes among the feeding sites (X^2 = 54.6, df = 5, p < 0.001). Flock sizes were larger at KD3 (141.1 ± 21.2 birds; n = 123) and FC1 (101.9 ± 11.9 birds; n = 164) but these sites acted as staging sites either side of low tide (Fig. 5.3; Fig. 5.8a). Flock sizes were smallest at FC5 (5.9 ± 1.3 birds; n = 20) and FC6 (18.6 ± 2.2 birds; n = 36) (Fig. 5.8a) in eastern Fullerton Cove. Overall, the proportion of foraging godwits was greater in flocks in the Fullerton Cove sites (sites 1, 2, 5 and 6) compared to the Kooragang dyke impoundments (sites 3 and 4) (X^2 = 113.5, df = 5, p < 0.001). Over 90% of godwits were foraging in flocks observed at the Fullerton Cove sites (Fig. 5.8b). The lowest percentage of foraging birds was observed for flocks at KD3 (53.5 ± 4.2%; n = 90), which is used as a staging site in late stage of the low tide period (Fig. 5.3). The percentage of foraging birds was high (> 80% of flock) from first exposure of mudflats at 2.25 hours before low tide until about 2.25 hours after low tide (Fig. 5.8b).

Foraging (X^2 = 63.5, df = 5, p < 0.001) and capture (X^2 = 58.3, df = 5, p < 0.001) rates of Bar-tailed Godwits differed significantly among survey sites. Capture rates were highest at site FC2 (7.0 ± 0.6 items min^{-1}) in Fullerton Cove and lowest at site KD4 at the Kooragang dykes (1.7 ± 0.3 items min^{-1}) (Table 5.5). Foraging rates did not differ among microhabitats (X^2 = 5.7, df = 3, p = 0.125) but Bar-tailed Godwits preferred to forage on wet mud or near the tide edge where their capture rates were highest (X^2 = 10.9, df = 3, p = 0.012). Although focal samples were limited, capture rates were lowest near the mangrove edge (Table 5.5). There was also evidence for age and sex-related effects on the foraging rates of Bar-tailed Godwits. Adult and juvenile birds had similar foraging rates ($F_{1,193}$ = 1.0, p = 0.313) but adult capture rates were almost double juvenile capture rates ($F_{1,193}$ = 17.6, p < 0.001) (Table 5.5). Female godwits made up 80% of focal samples but their foraging (10% lower) and capture (55% lower) rates differed significantly from male birds (foraging rate: $F_{1,176}$ = 4.6, p = 0.034; capture rate: $F_{1,176}$ = 6.1, p = 0.018) (Table 5.5). Nearest-neighbour distances for focal Bar-tailed Godwits were similar among the feeding sites (Table 5.5) and did not influence their capture rates (r = - 0.1, n = 361, p = 0.197).

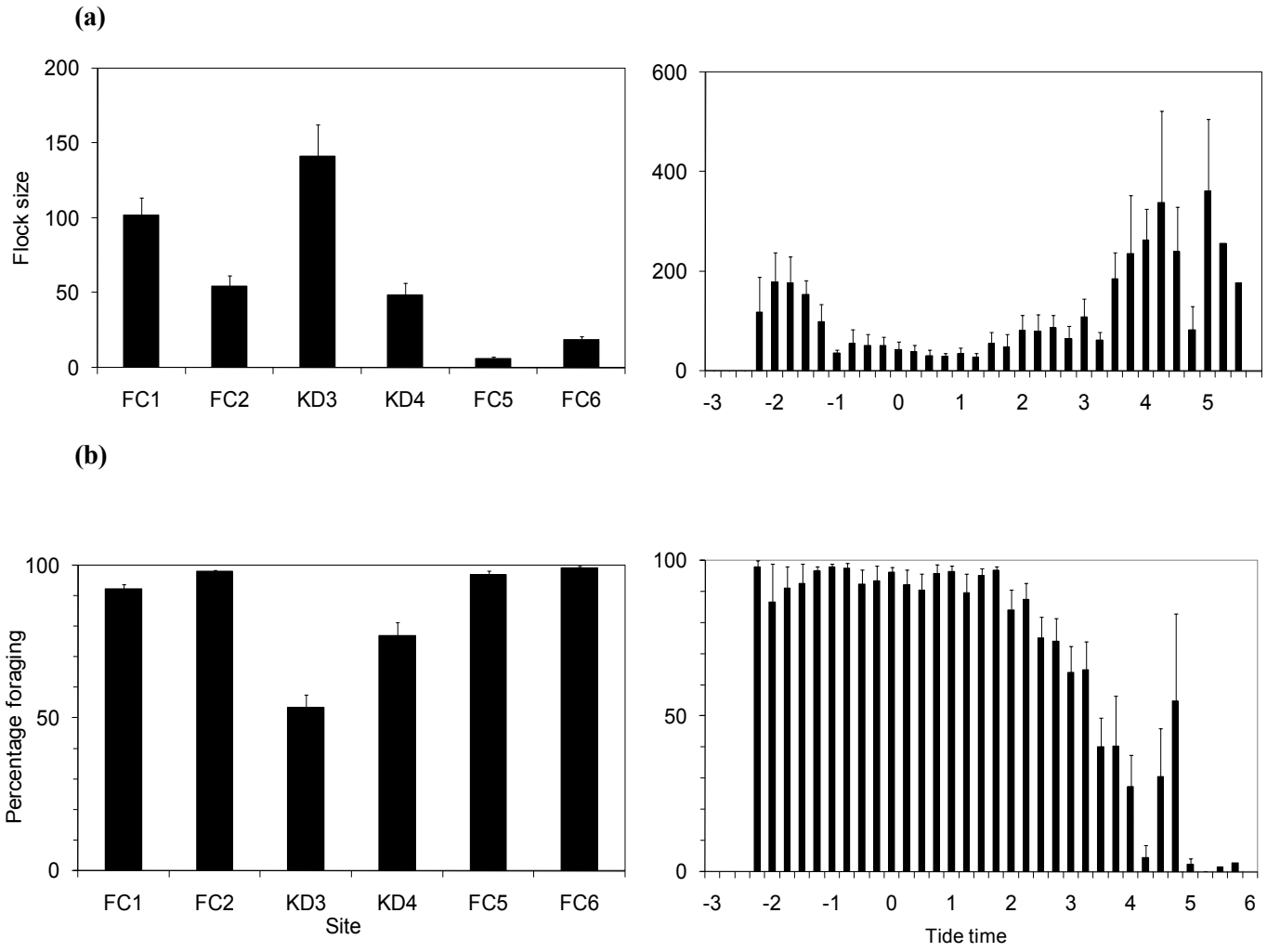


Figure 5.8 Mean (± S.E.) (a) flock sizes and (b) percentage of Bar-tailed Godwits foraging per flock scan in relation to feeding site and tide time (all sites combined). Mean values for sites are averaged over all flock scans (FC = Fullerton Cove sites, KD = Kooragang dyke sites). Mean values for tide time are averaged across all survey sites (tide time is measured tide height in hours (published low tide = 0) see Methods).

Table 5.5 Factors affecting mean (± S.E.) total foraging actions (pecks and probes) and capture rates (min^{-1}) of Bar-tailed Godwits in the Hunter estuary (n = number of focal samples; NND = mean nearest neighbour distance; FC = Fullerton Cove, KD = Kooragang dyke impoundments).

Factor		n	NND	Total pecks	Total probes	Total actions	Total captures
Site	FC1	71	10.3 (0.9)	39.3 (1.5)	19.2 (0.9)	58.5 (1.5)	3.3 (0.3)
	FC2	57	8.1 (0.8)	33.6 (2.2)	38.7 (2.8)	72.4 (2.1)	7.0 (0.6)
	KD3	117	6.7 (0.6)	37.3 (1.9)	22.6 (1.2)	59.7 (2.2)	3.8 (0.3)
	KD4	43	9.4 (1.2)	50.3 (2.6)	25.9 (1.6)	76.2 (2.6)	1.7 (0.3)
	FC5	30	9.5 (1.9)	35.5 (2.3)	21.9 (1.3)	57.4 (2.4)	2.7 (0.3)
	FC6	44	7.3 (0.7)	40.9 (1.2)	33.9 (1.7)	74.8 (1.8)	5.4 (0.9)
Microhabitat	Mudflat	194	8.8 (0.5)	36.8 (1.0)	26.7 (1.0)	63.5 (1.2)	4.5 (0.3)
	Mudflat/mangrove	10	8.0 (3.4)	31.4 (7.3)	29.3 (8.1)	60.7 (8.5)	2.7 (1.5)
	Mudflat/open water	36	9.7 (1.3)	32.9 (2.9)	29.5 (3.2)	62.3 (3.6)	4.3 (0.7)
	Open water	122	6.9 (0.6)	44.6 (1.7)	24.2 (1.1)	68.7 (1.9)	3.4 (0.3)
Age	Adult	118	8.4 (0.6)	42.1 (1.3)	29.3 (1.2)	71.4 (1.5)	4.6 (0.4)
	Juvenile	77	8.1 (0.7)	42.5 (1.8)	27.0 (1.3)	69.5 (2.0)	2.4 (0.2)
Sex	Male	36	6.6 (0.8)	42.1 (1.2)	34.4 (2.4)	76.6 (3.0)	6.3 (1.1)
	Female	142	9.5 (0.6)	41.6 (1.2)	27.4 (0.9)	69.1 (1.4)	2.8 (0.2)

5.5 Discussion

Many studies have linked the distributions of shorebird species to their preferred prey (e.g., Goss-Custard 1970; Ribeiro *et al.* 2004; Zharikov & Skilleter 2004c) but shorebird-prey relationships are not always this simple (Rogers 1999). In this study migratory shorebirds and benthic invertebrates were not evenly distributed across intertidal mudflats (Table 5.2; Fig. 5.5) and this may have been explained by the distribution of their preferred prey (Fig. 5.7). Shorebird distribution was strongly linked to mudflat availability within each tidal cycle (Fig. 5.4), but there were differences among species with most small shorebirds foraging in mudflats near the main day roosts, while larger Bar-tailed Godwits, Black-tailed Godwits, Eastern Curlews and Whimbrels often travelled further to feed on intertidal mudflats in Fullerton Cove. Generally, smaller shorebirds have greater energetic constraints than larger species and need to feed for longer periods (Zwarts *et al.* 1990b) (see Chapter 4). There was also a segregation of birds within Fullerton Cove, with most shorebirds using mudflats in the west (sites FC1 and FC2), while fewer birds used mudflats in the east (sites FC5 and FC6) (Table 5.2). A thick layer of oyster shells in south-eastern Fullerton Cove (FC5) may have made prey less accessible to foraging shorebirds.

There were also differences in benthic invertebrate assemblages among the survey sites, but my study only documented a snap-shot of their distributions which typically show a high degree of spatial and temporal variability (Jones *et al.* 1986; Dittmann 2002; Winberg *et al.* 2006). During surveys of Fullerton Cove from 1976-77, the Australian Littoral Society found that five benthic invertebrates dominated the intertidal mudflats, including two 'brooders' the bivalve *Arthritica helmsi* and the amphipod *Victoriopisa australiensis* which were abundant in these early surveys (Hutchings 1977, 1983) and during my sampling in 2006. Benthic invertebrates in Fullerton Cove often experience a large range of salinities, including prolonged periods of low salinity after high rainfall and runoff, which may explain the low species diversity and the high spatial variability seen in observed species (Hutchings 1983). In my study, sediment conductivity was highest in the Kooragang dyke impoundments (Table 5.4) which were not as well flushed as feeding sites in Fullerton Cove. Reduced tidal connectivity in the impoundments may have favoured the build up of organic matter and consequently high numbers of polychaete worms and the occasional formation of thick algal mats (Table 5.4; Fig. 5.5). Capture rates for Bar-tailed Godwits, however, were higher in mudflats in northern Fullerton Cove (FC2 and FC6) (Table 5.5) where polychaete worms, amphipods and small bivalves could be abundant (Fig. 5.5). Bar-

tailed Godwits are mostly carnivorous, often preying on soft-bodied invertebrates, molluscs and insects (Higgins & Davies 1996), but crabs are often important in their diet (Zharikov & Skilleter 2002, 2003a). Sentinel crabs *Macrophthalmus setosus* were a frequent prey item for Bar-tailed Godwits, Whimbrels and Eastern Curlews in this study (*pers. obs*.) and the latter two species were positively correlated with crab densities (Fig. 5.7b). Although, *Macrophthalmus setosus* were highly abundant on the mudflat surface of the feeding sites (*pers. obs*.) they were underestimated in the benthos sampling and burrow counts (Table 5.4).

Radio-tracking of Bar-tailed Godwits in the Hunter estuary showed that individuals used the same foraging locations, regardless of whether it was day or night (Richardson 2004; Foate 2005), with birds typically using the Kooragang dyke impoundments and mudflats near the entrance to Fullerton Cove as staging sites en route to their main feeding or roosting sites. Excluding the staging sites (FC1 and KD3), Eastern Curlews and Whimbrels were generally spread evenly across the feeding sites as these species are usually territorial foragers (Dann 1987; Higgins & Davies 1996; Zharikov & Skilleter 2004c). Bar-tailed Godwits were more gregarious, generally feeding in closely associated mixed flocks which showed few signs of aggressive behaviour. There are advantages to this flocking behaviour, as it usually increases overall flock vigilance (Metcalfe 1984b; Roberts 1996; Barbosa 1997), however, predator risk may not be an important issue for Bar-tailed Godwits in the Hunter estuary, as few birds of prey were seen at the feeding sites and only one successful attack by a Swamp Harrier was observed during the study (over 200 hours of observations). Within each feeding site Bar-tailed Godwits usually fed near the edge or in shallow water, usually away from fringing mangrove. Godwits feeding near the mangrove had the lowest capture rates overall (Table 5.5) but the mangroves may reduce their ability to detect predators, therefore, increasing vigilance levels and reducing feeding efficiency of foraging birds (Metcalfe 1984a).

There was also some evidence for differences in feeding behaviour among individual birds. More experienced adult godwits had capture rates double that of juveniles. Adults birds would also have been in hyperphagia during the time of sampling, and may have been foraging at a greater rate in order to prepare for their northward migration in the coming months. There was also some segregation between the sexes as males had greater capture rates than females (Table 5.5). This could have reflected differences in prey choice, as males may have taken more abundant but

smaller soft-bodied invertebrates while larger females can handle bigger prey, such as crabs, resulting in females capturing fewer prey per minute compared to males. This requires further study as other studies have also found sex-related differences in habitat use for this species with females usually dominating over males to select better quality feeding habitats (Taylor *et al.* 1999; Scheiffarth 2001; Zharikov & Skilleter 2002). In the Hunter estuary most focal godwits were females (Table 5.5) and this pattern has also been observed in other parts of south-eastern Australia (Barter 1989a; Nebel 2007). This has implications for species conservation as changes to habitats predominantly used by one sex or age group can have disproportionate effects on population size and the recruitment of second year birds (Nebel 2007).

One-third of the global Bar-tailed Godwit population migrate to spend their non-breeding season in Australia each year (Wilson *et al.* 2007) and depend on good quality feeding habitat in order to refuel (Landys-Ciannelli *et al.* 2003) for their return migrations to Alaska and Siberia (Gill *et al.* 2008). Understanding differences in the distribution of shorebirds and their prey are critical if we are to predict the effect of environmental change on a given population. Some shorebird species can be particularly vulnerable to habitat changes and associated effects on their prey (West *et al.* 2007), especially when populations depend on a single prey species (e.g., Baker *et al.* 2004; Van Gils *et al.* 2006). Invertebrate stocks in the Hunter estuary are threatened by further port development and water extraction upstream, which may change tidal regimes, and reduce freshwater and nutrient inputs. Increased shipping activity will also increase the risk of introducing exotic pests in ballast waters. It is difficult to predict how shorebirds will respond to further loss of intertidal habitats in the Hunter estuary but other studies have shown that habitat degradation can directly impact mortality and survival rates of shorebirds (Durell *et al.* 2005; Goss-Custard *et al*. 2006). Further modification of the Hunter estuary will add to the cumulative loss of non-breeding habitat in south-eastern Australia (Nebel *et al.* 2008) and in stop-over sites in south-east Asia (Barter 2002; Moores *et al.* 2008). In light of ongoing threats to shorebird habitats throughout the EAAF it is critical to ensure adequate provision and management of intertidal mudflats in south-eastern Australia to support foraging shorebirds during their non-breeding seasons.

CHAPTER 6: FEEDING BEHAVIOUR OF SHARP-TAILED SANDPIPERS *CALIDRIS ACUMINATA* IN COASTAL SALTMARSH

6.1 Abstract

Coastal saltmarsh was listed as an Endangered Ecological Community (EEC) in three coastal bioregions in New South Wales (NSW), south-eastern Australia (*NSW Threatened Species Conservation Act 1995*), following widespread declines and concerns that alteration of tidal regimes and disturbance from coastal developments were causing further fragmentation. In its final determination, the NSW Scientific Committee cited the importance of saltmarsh for migratory shorebirds, however, few studies have documented this in detail. In this chapter, I investigated feeding behaviour of a common migratory shorebird, the Sharp-tailed Sandpiper *Calidris acuminata*, in saltmarsh in the Hunter estuary, a significant non-breeding site in NSW. Feeding behaviour dominated flock behaviour (69 ± 3%), regardless of the effects of tidal period. Time of day did influence feeding behaviour, with more birds feeding during midday compared to early morning or evening surveys. In focal birds, prey capture rates in saltmarsh were highest at high tide (4.9 ± 0.4 captures min^{-1}) and in open water microhabitat (5.8 ± 0.7 captures min^{-1}). Overall, capture rates for Sharp-tailed Sandpipers in saltmarsh were twice that of Curlew Sandpipers *Calidris ferruginea,* a similar-sized shorebird, feeding in nearby intertidal mudflats, however, these differences were most likely due to differences in prey choice. Adequate protection and rehabilitation of coastal saltmarsh should be a key consideration for managers of shorebird habitat in south-eastern Australia. Significant areas of saltmarsh in the Hunter estuary are outside of the reserve system but require active management to control mangrove encroachment and disturbance from recreational vehicles. They are also under serious threat of reclamation and infilling to support industrial expansion for the nearby city of Newcastle.

6.2 Introduction

Worldwide, many coastal wetlands have been impacted by development and rapid human population growth (Dugan 1993). In Australia, large areas of saltmarsh have historically been drained for agricultural and urban development (Adam 1981; Bucher & Saenger 1991; Zann 1995) and remaining saltmarsh is threatened by ongoing development pressures and insensitive use (Adam 2002; Laegdsgaard 2006). In temperate Australia, coastal saltmarshes are characterised by halophytic herbaceous and low woody vascular plants in the upper intertidal zone (between mean high water spring and neap tides), typically upslope of large stands of mangroves (Adam 1981). Altered tidal regimes in parts of south-eastern Australia have favoured mangrove expansion upslope into saltmarsh (Saintilan & Williams 2000; Saintilan & Wilton 2001), with 70% of estuaries experiencing significant losses (> 30%) of saltmarsh due to mangrove encroachment and greatest declines seen in the most developed estuaries (Saintilan & Williams 1999).

In 2004, coastal saltmarsh was listed as an Ecologically Endangered Community (EEC) in the New South Wales (NSW) North Coast, Sydney Basin and South-East Corner bioregions (*NSW Threatened Species Conservation Act 1995*), by a scientific committee which cited the significance of saltmarsh for estuarine fauna, including migratory shorebird species (DECC 2004). At least nine migratory shorebird species regularly feed and/or roost in Australian saltmarshes (Spencer *et al.* 2009), typically open areas where shorebirds can detect predators easily. Most migratory shorebirds found in Australia migrate to Alaska, Siberia, Mongolia, northern China or Japan (June-July) to breed each year. They can use many habitats in Australia during their non-breeding season (Sept-Apr), including coastal wetlands, where they typically feed at low tide and roost at high tide (Lane 1987).

Saltmarsh declines may have directly impacted migratory shorebird populations by reducing the availability of feeding habitat, however, little is known of the relative importance of this habitat for shorebird species. Much of our current knowledge of feeding ecology of calidridine shorebirds in saltmarshes is from outside Australia including: South Africa (e.g., Kalejta 1992; Velasquez & Hockey 1992), North America (e.g., Bildstein *et al.* 1982; Erwin 1996) and Europe (e.g., Goss-Custard & Yates 1992; Norris 2000; Rosa *et al.* 2003), where some shorebirds fed in mudflats at low tide and switched to saltmarshes to continue feeding through high tide (Puttick 1979; Yasue *et al.* 2003).

In this chapter, I investigated the feeding behaviour of Sharp-tailed Sandpipers *Calidris acuminata* in saltmarsh habitats in relation to the tidal period, time of day and microhabitat in the Hunter estuary, south-eastern Australia (2005-06). Sharp-tailed Sandpipers are common in saltmarsh (Higgins & Davies 1996) and can reach internationally significant numbers (> 1% of their flyway population) in the Hunter estuary during the austral summer (see Chapter 2). I compared Sharp-tailed Sandpiper feeding behaviour to a similar sized species, the Curlew Sandpiper *C. ferruginea*, which fed on nearby intertidal mudflats, and I also compared prey availability between these two habitats.

6.3 Methods

6.3.1 Study site

The Hunter estuary (32° 51'S/ 151° 46'E) borders the large city of Newcastle in NSW, Australia (Fig. 6.1). Development associated with this major port has resulted in the loss of at least 67% (1,428 ha) of saltmarsh in the Hunter estuary through the draining of land for industrial and agricultural use (Williams *et al.* 2000). Kooragang Island (Fig. 6.1), a major feature of the estuary, was formed following the amalgamation of seven of the original islands by dredging works from 1845-1968 (see Chapter 2). The southern section of Kooragang Island is now used for heavy industry (principally coal export), but its northern and western portions contain areas of mangrove and saltmarsh which are managed by the Kooragang Wetland Rehabilitation Project (KWRP) and the NSW Department of Environment and Climate Change. The Hunter estuary still contains significant areas of saltmarsh (705 ha), mangrove (1,711 ha) (Williams *et al.* 2000) and intertidal mudflats (>900 ha) (Geering 1995) which attract large numbers of migratory shorebirds to the estuary each year (Smith 1991). During summer months (Sept-Apr), up to 35 migratory shorebird species can be seen in the estuary, with seven species in nationally significant numbers (> 1% Australian population) (see Chapter 2). The Hunter estuary is listed as an internationally significant wetland under the Ramsar Convention (RIS 2002) and the East Asian-Australasian Shorebird Site Network and 4,255 ha are formally protected under the Hunter Wetlands National Park (Fig. 6.1) (DECC 2007).

The main study site was an area of saltmarsh on the western side of Kooragang Island, known as Area E (124 ha) (Fig. 6.1), which is being rehabilitated by the KWRP (Svoboda 1996). This saltmarsh consists of shallow tidal pools fringed by *Sporobolus virginicus* and *Sarcocornia quinqueflora* (Fig. 6.2) (Outhred & Buckney 1983), which are flooded periodically by spring high tides and can support up to 20 species of migratory shorebirds, including large numbers of Sharp-tailed Sandpipers (up to 1,711 birds) (see Chapter 7). Foraging Curlew Sandpipers were observed on an artificial intertidal mudflat (8.5 ha), impounded by a training wall in the main channel of the Hunter River (Fig. 6.1). Exposure of this mudflat lags behind natural mudflats in the rest of the estuary by about two hours and is exposed from published low tide to about one hour before published high tide (five to six hours per cycle), providing feeding habitat for at least 14 migratory shorebird species (see Chapter 4). High tide heights ranged from 1.13 - 1.88 m and low tides from 0.22 - 0.66 m during this study (BOM 2004).

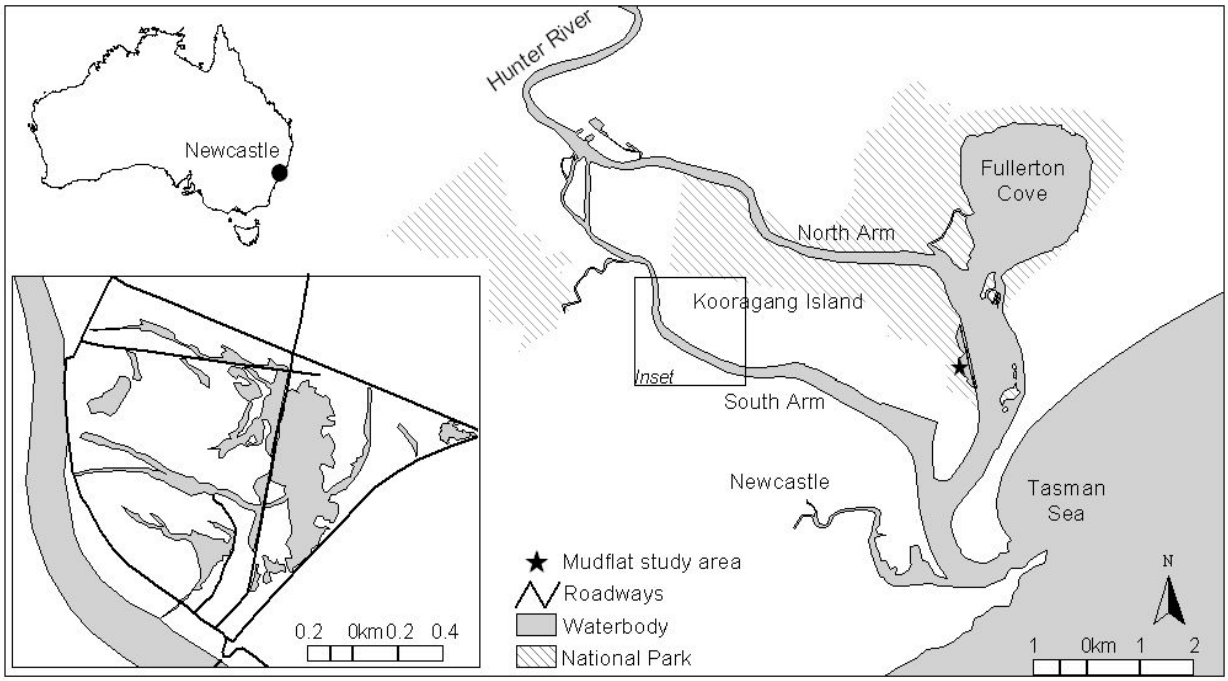


Figure 6.1 Location of the Hunter estuary, south-eastern Australia, and the main study area (Area E) on western Kooragang Island (inset). The intertidal mudflat was located on the North Arm of the Hunter River (indicated).

Credit: J. Spencer

Figure 6.2 A Sharp-tailed Sandpiper *Calidris acuminata* foraging in flooded saltmarsh (left) in the main study area, Area E (right), on Kooragang Island, in the Hunter estuary.

6.3.2 Flock scans

To investigate Sharp-tailed Sandpiper feeding behaviour in saltmarsh, I observed flocks in the main study area from January to March 2005 (33 hours, 104 flock scans, N = 15). I randomly assigned sampling dates but stratified sampling by tidal period (high (N = 9); low (N = 6)) and time of day (morning (0630 - 0930; N = 7); midday (1100 - 1400; N = 5); and evening (1700 - 1930; N = 3)). Additional surveys (during flood tides only) were done in the 2006 non-breeding season (Feb-Mar; N = 5) to investigate differences in the feeding behaviour of Sharp-tailed Sandpipers in saltmarsh and Curlew Sandpipers in intertidal mudflat. Although both species were observed in mixed flocks in both habitats, few Sharp-tailed Sandpipers fed in intertidal mudflat (see Chapter 4) and Curlew Sandpipers were uncommon in saltmarsh (see Chapter 7). Flocks were observed over a two to three hour period using a 20 – 60X zoom telescope at distances of 5 - 150 m, from an access road that traverses the saltmarsh and from a rock wall that borders the intertidal mudflat (Fig. 6.1). Instantaneous flock scans (Altmann 1974) were completed every 30 minutes, during which the behaviour of each bird in the flock was dictated onto a microcassette recorder. Behavioural categories included; feeding (actively pecking/probing the substrate or handling prey), resting (head tucked in scapular feathers behind shoulders), preening (cleaning plumage and bathing), and other (loafing, walking, aggression, vigilance and flying).

6.3.3 Feeding rates

Actively feeding Sharp-tailed Sandpipers were selected using a random number table (Altmann 1974) and observed for two minute periods (n = 174) in the 2005 season. Additional focal observations (one minute samples) were made for Sharp-tailed Sandpipers (n = 34) and Curlew Sandpipers (n = 47) in the 2006 season. All feeding activity, microhabitat, nearest-neighbour distance (relative to bird length) and species were dictated onto a tape recorder for each focal bird and transcribed later. Feeding activity included: pecks (touch of the substrate/water surface); probes (insertion of bill into substrate/below water surface); and captures (swallowing action or clear view of prey item). There were five types of microhabitat in the saltmarsh study area: saltmarsh vegetation; open water; saltmarsh/open water; saltmarsh/mudflat; open water/mudflat (the latter three being transition zones). Incomplete observations, where birds changed behaviour as a result of disturbance or took flight, were discarded. Two measures of feeding behaviour were calculated: (i) a foraging rate (the sum of all bill movements (pecks and probes) per minute); and (ii) a capture rate (the total number of captures (or swallows) per minute).

6.3.4 Invertebrate sampling

To determine prey availability for focal species, I sampled benthic invertebrates in the saltmarsh and intertidal mudflat study sites in January 2006. Three plots (about 50 m apart, 5 m radius) were sampled in both study sites, with a PVC corer (90 mm diameter) driven into the mud to a depth of 50 mm to collect six cores per plot. This depth represented the fraction of benthos accessible to Sharp-tailed Sandpipers and Curlew Sandpipers, their bill lengths ranging from 28 to 44 mm respectively (Higgins & Davies 1996). Samples were sieved on site in a 500 µm open-ended mesh bag (430 x 270 mm), preserved in 7% formalin and later transferred to 70% ethanol. Invertebrates were sorted to family level only. In saltmarsh, I also collected above-ground invertebrates by sweep-netting for ten minutes in a 5 m radius around each plot, freezing samples to sort later. I collected fresh faeces near actively-feeding Sharp-tailed Sandpiper in saltmarsh and examined these samples under a microscope (40x magnification). Undigested chitinous insect parts made up the bulk of these faecal samples but this was not quantified and is not presented in this chapter.

6.3.5 Statistical analyses

Flock percentage data had non-normal distributions, even after transformation with the arcsine function (Fowler *et al.* 1998), therefore, I used non-parametric Kruskal-Wallis tests, comparing the *H*-statistic to chi-square distribution (Quinn & Keough 2002), to investigate the effects of tidal period and time of day on the proportion of feeding, resting and preening behaviour in Sharp-tailed Sandpiper flocks. Estimation of the interaction term in a multifactor Analysis of Variance (ANOVA) can be unreliable when sample sizes are unequal (Underwood 1997), therefore, I used one-way ANOVA tests (with Tukey HSD *post hoc* comparisons) (Quinn & Keough 2002) to determine whether foraging and capture rates differed between tidal periods or among times of day and microhabitats. Foraging and capture rates were transformed with the square root function to meet the assumptions of tests for normality (Kolmogorov-Smirnov's) and equality of variances (Levenes) (Fowler *et al.* 1998). I excluded focal observations in saltmarsh vegetation (n = 5) from the microhabitat analysis, as the vegetation prevented accurate measurement of feeding behaviour in focal birds. I used single factor ANOVA tests to determine if ($\log_{10}$ transformed) foraging or capture rates differed between species or if total benthic invertebrate abundance (after $\log_{10}$ transformation) differed between the study sites. Statistical analyses were carried out in SPSS (2005). All values are given as means (± standard error).

6.4 Results

6.4.1 Flock behaviour

Sharp-tailed Sandpipers fed in loosely spaced flocks in saltmarsh habitat (mean nearest neighbour distance 7.7 ± 0.6 bird lengths, range 1 - 60; mean flock size 97.9 ± 14.2 birds, range 12 - 873) and were neighbours of 86% of focal birds (other species included: Black-winged Stilts *Himantopus himantopus* (10%); Marsh Sandpipers *Tringa stagnatilis* (2%); Red-necked Stints *Calidris ruficollis* (1%); Curlew Sandpipers (0.5%); and Masked Lapwings *Vanellus miles* (0.5%)). In the 2005 season, most Sharp-tailed Sandpipers were feeding (69.3 ± 2.5%), while few were observed resting (1.5 ± 0.5%), preening (9.8 ± 1.0%), or in other (19.4 ± 2.3%) behaviours (Fig. 6.3). This pattern was consistent between years and species, with a high feeding percentage for Sharp-tailed Sandpipers in saltmarsh habitat (69.4 ± 13.3%, n = 8) and Curlew Sandpipers in intertidal mudflat (70.5 ± 6.7%, n = 14) in the 2006 surveys. Although the tidal period did not affect the behaviour of Sharp-tailed Sandpiper flocks (feeding: $X^2 = 3.2$, df = 1, $p = 0.075$; resting: $X^2 = 0.7$, df = 1, $p = 0.416$; preening: $X^2 = 0.2$, df = 1, $p = 0.686$) (high, n = 65; low, n = 39), time of day did affect levels of feeding ($X^2 = 15.9$, df = 2, $p < 0.001$) and preening ($X^2 = 8.2$, df = 2, $p = 0.017$). A higher percentage of Sharp-tailed Sandpipers fed at midday (84.3 ± 3.3%; n = 36), compared to morning (59.9 ± 4.3%; n = 44) or evening (63.9 ± 6.1%; n = 24), while most birds preened in the morning. Resting behaviour was uncommon (0.1-2.4%) in all time periods ($X^2 = 5.6$, df = 2, $p = 0.061$) (Fig. 6.3).

6.4.2 Feeding rates and prey availability

Tidal period ($F_{1,172} = 0.01$, $p = 0.978$), time of day ($F_{2,171} = 1.4$, $p = 0.243$) and microhabitat ($F_{4,169} = 2.3$, $p = 0.082$) had no effect on the foraging rates (actions min^{-1}) of Sharp-tailed Sandpipers in saltmarsh habitat (Table 6.1). Time of day also had no effect on capture rates ($F_{2,171} = 0.4$, $p = 0.676$) but birds were most successful during high tide periods ($F_{1,172} = 5.8$, $p = 0.018$) and in or near the edge of open water ($F_{4,169} = 4.0$, $p = 0.009$) (Table 6.1). Overall, Curlew Sandpipers foraged faster (78.1 ± 3.8 actions min^{-1}) than Sharp-tailed Sandpipers (55.3 ± 3.2 actions min^{-1}) ($F_{1,79} = 22.9$, $p < 0.001$) but capture rates for Sharp-tailed Sandpipers in saltmarsh habitat (4.9 ± 0.5 captures min^{-1}) were more than double that of Curlew Sandpipers in intertidal mudflat (2.3 ± 0.3 captures min^{-1}) ($F_{1,79} = 24.6$, $p < 0.001$). Total benthic invertebrate abundance did not differ between the saltmarsh (18.0 ± 5.1 individuals $core^{-1}$) and intertidal mudflat (25.4 ± 9.1 individuals $core^{-1}$) study areas ($F_{1,34} = 0.4$, $p = 0.555$) but large numbers of small bivalves

(Leptonidae) and large polychaete worms (Orbiniidae and Nereididae) were collected from the intertidal mudflat (Fig. 6.4). Above-ground invertebrate density could be high in saltmarsh, as large numbers of adult non-biting midges (Diptera: Chironomidae) (32.0 ± 20.9 individuals per m^2; range 8.3 – 73.7) were caught during sampling. Other diptera (0.03 ± 0.01 per m^2) and arachnids (0.1 ± 0.02 per m^2) made up the rest of samples.

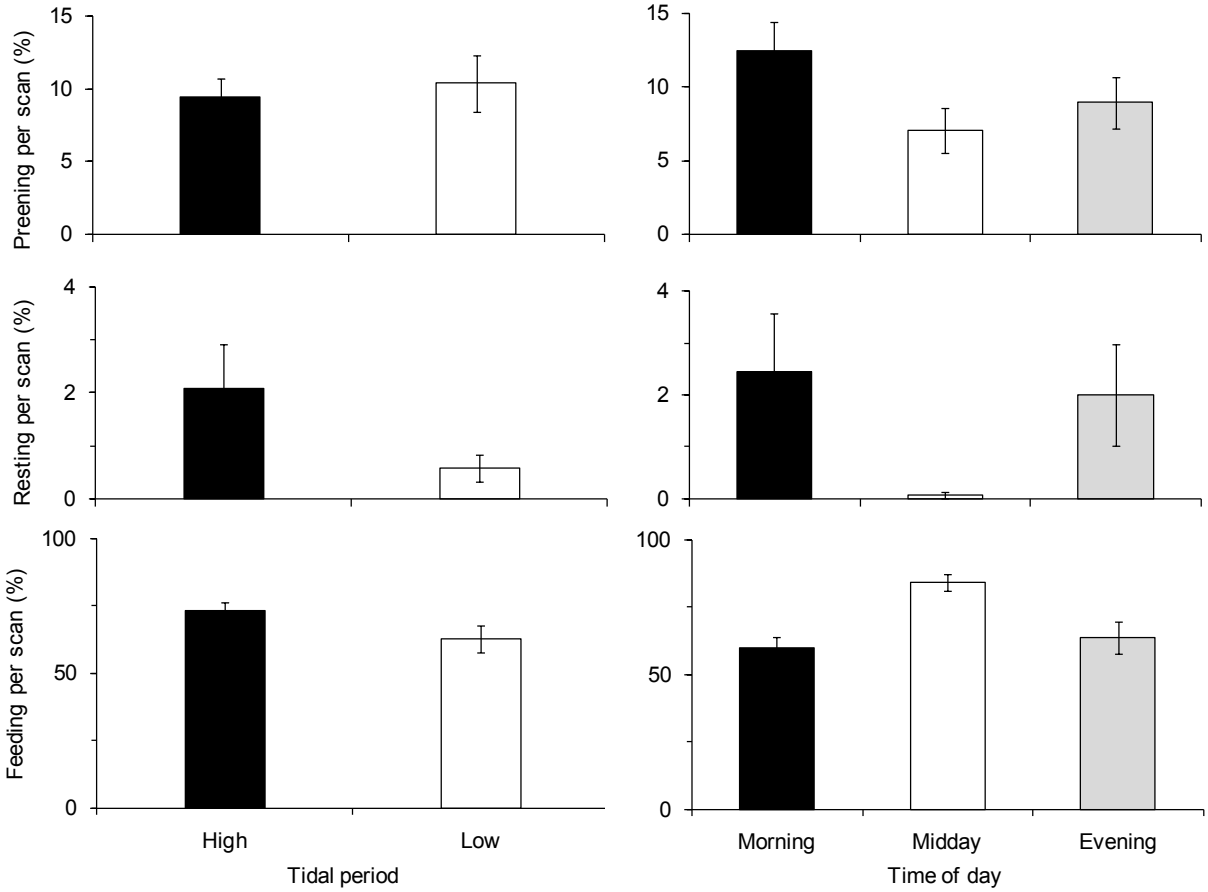


Figure 6.3 Mean (± S.E.) percentage of feeding, resting and preening behaviour observed in Sharp-tailed Sandpiper flocks in saltmarsh habitat during different tidal periods and times of day (Jan-Mar 2005; n = 104).

Table 6.1 Summary of Sharp-tailed Sandpiper foraging activity (min^{-1}), including mean (± S.E.) bill movements (pecks, probes and total actions) and captures in saltmarsh habitat, in relation to tidal period, time of day and microhabitat (*denotes significant differences within factors; n = number of focal samples).

Factor		n	Pecks	Probes	Total actions	Captures
Tide	High	100	27.5 (1.8)	15.6 (2.2)	43.1 (2.0)	4.9 (0.4)*
	Low	74	36.5 (2.6)	7.7 (2.0)	44.3 (2.8)	3.6 (0.3)*
Time	Morning	70	31.4 (2.7)	15.9 (3.1)	47.3 (3.0)	4.2 (0.4)
	Midday	56	34.0 (2.3)	6.8 (1.9)	40.8 (2.5)	4.7 (0.5)
	Evening	48	28.1 (2.8)	13.3 (2.3)	41.4 (3.0)	4.4 (0.4)
Microhabitat	Saltmarsh	5	34.6 (11.4)	26.1 (15.7)	60.7 (9.7)	2.0 (0.8)
	Open water	40	26.0 (3.3)	17.5 (2.9)	43.5 (3.2)	5.8 (0.7)*
	Saltmarsh/open water	56	27.3 (2.1)	10.2 (2.1)	37.5 (2.6)	4.7 (0.5)
	Saltmarsh/mudflat	61	36.8 (2.6)	10.1 (3.2)	47.0 (3.0)	3.4 (0.3)*
	Open water/mudflat	12	38.6 (7.3)	9.3 (3.1)	47.9 (6.8)	4.4 (0.9)
Total		174	31.3 (1.5)	12.3 (1.5)	43.6 (1.7)	4.4 (0.3)

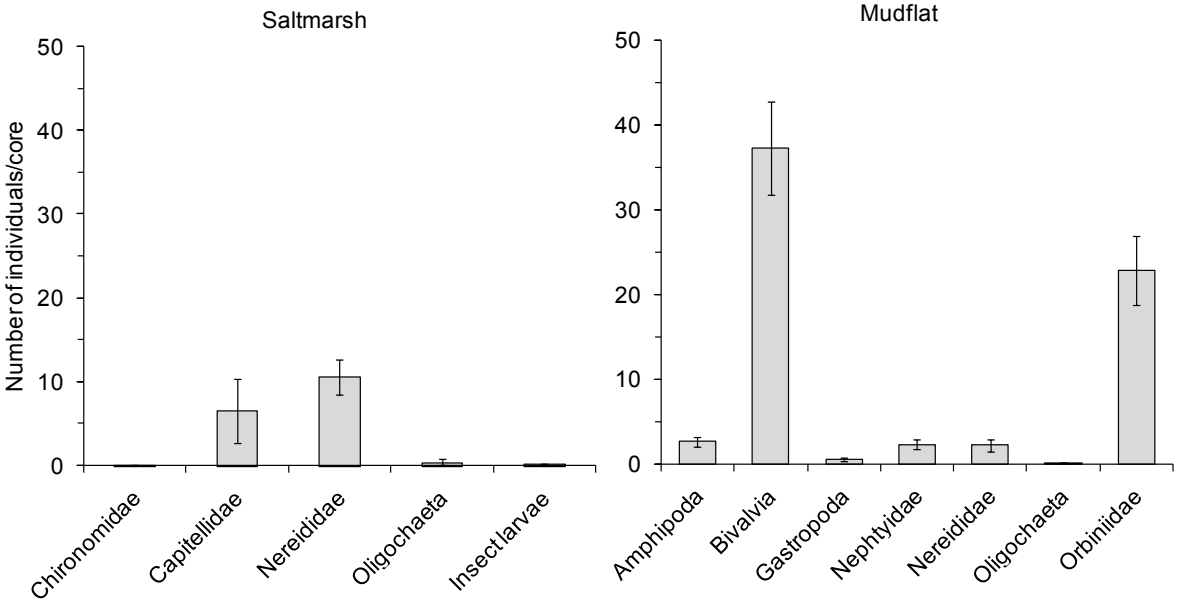


Figure 6.4 Mean (± S.E.) number of benthic invertebrates in saltmarsh and intertidal mudflat study sites in the Hunter estuary, NSW (January 2006).

6.5 Discussion

Overall, feeding dominated Sharp-tailed Sandpiper behaviour in saltmarsh, regardless of tidal period and time of day. Sharp-tailed Sandpipers fed nearly all day, resting only minimally, even during high tide periods (Fig. 6.3). Often the same flocks were present in the study area throughout the day and few Sharp-tailed Sandpipers appeared to move into or out of the saltmarsh in response to intertidal mudflat availability in the Hunter River. In reality, Sharp-tailed Sandpipers were only occasional visitors to mudflats in the Hunter River, being seen in only 24% of flock scans (maximum flock sizes 14-19 birds) (see Chapters 4 and 5).

Reduced tidal connectivity in the saltmarsh compared to the intertidal mudflats meant that water depth, and consequently feeding habitat availability, remained relatively constant in the saltmarsh in a given day. Some shorebirds respond to reduced feeding habitat availability by feeding in supratidal habitats (including saltmarshes) during high tide periods or remaining in these habitats throughout the tidal cycle (Dann 1999b; Masero *et al.* 2000; Masero & Perez-Hurtado 2001). Flocks of Sharp-tailed Sandpipers could be large (1,120.3 ± 279.2 birds) and Common Greenshanks *Tringa nebularia*, Marsh Sandpipers *Tringa stagnatilis* and Red-necked Stints *Calidris ruficollis* were also observed feeding in the study area.

As well as dictating where and when shorebird species can feed, the tidal cycle also determines the types, sizes and quantities of food that are available (Dann 1987; van de Kam *et al.* 2004). In saltmarsh, Sharp-tailed Sandpipers had almost unlimited access to benthic and terrestrial invertebrates, but Curlew Sandpipers could only feed on benthic invertebrates while intertidal mudflats were exposed. Overall, Sharp-tailed Sandpipers' capture rates were double that of Curlew Sandpipers in intertidal mudflat. This may be due to greater prey accessibility in saltmarsh, where adult chironomids were very active in the middle of the day (*pers. obs.*). Chironomid larvae are common in the diets of many shorebird species (Rehfisch 1994; Sherfy *et al.* 2000; Sanchez *et al.* 2006) including Sharp-tailed Sandpipers, which are highly omnivorous (Higgins & Davies 1996). Examination of Sharp-tailed Sandpipers faeces in this study confirmed that adult chironomids were an important dietary item for some birds and high densities of adult chironomids (up to 74 individuals per m^2) could be observed in the saltmarsh study area. These insects would be easy to detect on the water surface, enabling higher capture rates relative to foraging effort compared to foraging Curlew Sandpipers which usually target polychaetes buried

in the sediment (Thomas & Dartnall 1971; Dann 1999a). Microhabitat also influenced capture rates in Sharp-tailed Sandpipers with higher rates in open water, where birds were able to take adult insects (chironomids) from the water surface.

Coastal saltmarshes also provide night roosting habitat for many shorebird species (see Chapter 3), including the Sharp-tailed Sandpiper (*pers. obs.*). By feeding in saltmarsh throughout the day Sharp-tailed Sandpipers reduce the need to expend extra energy travelling to and from separate day and night roosts. Although many shorebirds roost in the North Arm of the Hunter River during day time high tides (see Chapter 3), most species select different sites to roost in at night (Handel & Gill 1992; Rogers 2006), including flooded saltmarsh, which can be hard for ground predators to access.

Saltmarsh habitats provide additional benefits to Sharp-tailed Sandpipers by reducing their predation risk. Raptors can be a major source of mortality in overwintering shorebirds (Page & Whitacre 1975; Whitfield 2003b). Diurnal raptors were observed regularly in this study, with at least 20 species recorded in the Hunter region (Stuart 2006, 2007). Sharp-tailed Sandpipers in non-breeding plumage (Fig. 6.2) are well camouflaged in saltmarsh vegetation and in this study were observed flattening themselves against saltmarsh vegetation when raptors were overhead or if alarm calls were given by other members of the flock.

Although Sharp-tailed Sandpipers are one of the most common migratory shorebird species in south-eastern Australia, declines observed in some areas have coincided with saltmarshes lost to coastal developments (Lane 1987). Sharp-tailed Sandpipers also regularly frequent freshwater wetlands in inland Australia (Higgins & Davies 1996) and so may have been impacted by severe drought and river regulation, and its associated effects on wetland health and extent in the Murray-Darling Basin (Kingsford 2000; Kingsford & Thomas 2004). Coastal saltmarshes provide significant feeding, roosting and staging habitat for this species during their non-breeding season and so have the potential to buffer populations from the effects of habitat loss in Australia and other parts of the flyway.

On Kooragang Island, areas of saltmarsh outside of the Hunter Wetlands National Park (Fig. 6.1) are managed by the KWRP which have undertaken on-ground works to create and rehabilitate

saltmarsh for shorebirds since 1993 (see Chapter 7). The KWRP manage the most immediate threats to saltmarsh, including insensitive use by recreational vehicles and mangrove encroachment, but the main study area is also zoned under NSW planning policies (*State Environment Planning Policy (SEPP) 74* and *SEPP (Major Projects)* which designate this site for industrial use. These policies may be put into effect under a new 20 year development plan to expand shipping facilities in Newcastle port (NPC 2009).

This study highlights the importance of coastal saltmarsh as feeding habitat for Sharp-tailed Sandpipers. The protection and restoration of these habitats is important for the successful migration and survival of this species. Water level management is the key to restoring and maintaining suitable areas of saltmarsh, with shallow ponds and short, sparse vegetation, for shorebirds. Further studies of coastal saltmarshes are needed to determine the importance of this habitat for other shorebird species and to provide information for management of water levels so that habitat availability and invertebrate biomass can be maximised for shorebirds during their non-breeding seasons.

CHAPTER 7: REHABILITATING WETLAND HABITAT FOR MIGRATORY SHOREBIRDS

7.1 Abstract

Wetland rehabilitation is a growing discipline, responding to concerns over global declines in wetland area and associated losses in ecosystem services and biodiversity. The principal goal of habitat rehabilitation is to return a damaged or degraded habitat to a fully functional ecosystem, irrespective of its original condition, and in the case of wetlands can include the removal of artificial structures or encroaching vegetation. In the Hunter estuary, south-eastern Australia, culverts were removed in 1995 to improve tidal flushing into wetlands on Ash Island, which has been heavily modified by the development of the port of Newcastle. The main aim was to rehabilitate wetland vegetation and habitat for waterbird and fish populations. Long-term monitoring (1994 - 2006) indicated tidal reinstatement had indirectly reduced habitat availability for migratory shorebirds by promoting the expansion of mangroves. Most shorebirds avoided wetlands dominated by mangroves, preferring sites with open saltmarsh and tidal pools. Total numbers of migratory shorebirds, however, were highly variable among years, which may have reflected local and regional changes in wetland availability, chance sightings of staging species, variations in their breeding success in the high Arctic, and loss of wetland habitat along the East Asian-Australasian flyway. Further rehabilitation work was carried out in 2005-06, where mangroves were removed in three areas formerly used by shorebirds, but there was limited opportunity to detect significant short-term impacts on shorebird distribution as total numbers of shorebirds were low in the following non-breeding season.

7.2 Introduction

Wetland loss and degradation is widespread and the greatest threat to migratory shorebird populations in Australia (DEH 2005). Australia is a highly urbanised country with more than 80% of its population living within 100 km from the coast (NLWRA 2002). In New South Wales (NSW), Australia's most populous state, at least 60% of coastal wetlands have been destroyed through draining and clearing for agricultural, urban and industrial development (Bowen *et al.* 1995). These developments have often disturbed or destroyed shorebird roosting and feeding habitat (Kingsford & Ferster Levy 1997; Blumstein *et al.* 2003).

Despite continued pressure on coastal wetlands, there is great potential to rehabilitate degraded estuarine habitats through the removal or manipulation of engineered structures (Williams & Watford 1997). Attitudes towards wetland management have also changed with increasing awareness of the importance of coastal wetlands (Streever *et al.* 1998) for sea defence, floodwater absorption, fisheries production and biodiversity (Mitsch & Gosselink 2000). In north-western Europe, for example, management responses to predicted sea-level rise include large-scale de-embankment projects, where intertidal habitats have been reinstated to protect low-lying flood-prone settlements (Wolters *et al.* 2005; Garbutt *et al.* 2006; Rupp-Armstrong & Nicholls 2007), benefiting some shorebird species (Atkinson *et al.* 2004). Most shorebird habitat rehabilitation projects, however, focus on high tide roost creation (Lawler 1995; Burton *et al.* 1996; Harding *et al.* 1999) as these habitats are often most heavily impacted by disturbance and development (Burton *et al.* 1996; Rogers *et al.* 2006b), but there have also been attempts to create or enhance feeding habitat (Rehfisch 1994; Evans *et al.* 1998; Sanders 2000; Lindegarth & Chapman 2001) and saltmarshes (Burchett *et al.* 1998; Atkinson 2003; Nelson 2006) for shorebird species in the United States, Europe, Japan and Australia.

Government support for wetland conservation has increased in Australia, with the development of state and federal policies (e.g., *NSW Wetlands Management Policy 1996*) that encourage the rehabilitation of wetlands. There are at least 69 wetland rehabilitation projects (covering almost 150,000 ha) in Australia (Streever 1997) and one of the largest is the Kooragang Wetland Rehabilitation Project (KWRP) (Streever 1998), initiated in 1993 to manage and rehabilitate wetlands in the Hunter estuary, NSW (Fig. 7.1). The Hunter estuary provides non-breeding habitat for migratory shorebirds (Smith 1991; Watkins 1993), but has suffered significant

reductions in estuarine habitats (Kingsford & Ferster Levy 1997; Williams *et al.* 2000) following the development of one of the world's most important coal export ports.

The KWRP is rehabilitating three wetland areas in the Hunter estuary: Stockton sandspit (10 ha); Ash Island (780 ha); and Tomago Island (800 ha) (Fig. 7.1) (Svoboda 1996). Extensive on-ground works at Stockton sandspit, including the excavation of a large lagoon, weir construction (to control tidal flushing) and the removal of terrestrial weeds and mature mangroves (Geering & Winning 1994), have successfully rehabilitated high tide roosting habitat at this site. The KWRP has also conducted on-ground works to rehabilitate and create saltmarsh on Ash Island. In 1995, the KWRP removed culverts in two creeks to improve tidal flushing on Ash Island, with the aim of restoring wetland vegetation and habitat for fish and waterbirds. There have also been several periods of mangrove removal to improve habitat for shorebirds, as most species prefer habitat with an open aspect (Lawler 1996; Rogers *et al.* 2006a) and avoid tall vegetation, including mangroves, which can hide predators (Dekker 1998; Dekker & Ydenberg 2004).

A common criticism of many rehabilitation programs, is the lack of long-term monitoring to measure the ecological success of rehabilitation efforts (Field 1998; Grayson *et al.* 1999; Zedler 2000). However, the removal of culverts on Ash Island was timed so that studies of vegetation (Streever *et al.* 1996; MacDonald 2001; Nelson 2006), fish and decapods (Williams *et al.* 1995), and waterbirds (Kingsford *et al.* 1998) were carried out before and after tidal manipulation. Comparisons before and after culvert removal detected a shift in the distribution of estuarine vegetation, with an expansion of mangrove and saltmarsh and a decline in terrestrial communities (MacDonald 2001; Nelson 2006; Howe 2008) (see section 7.3.1).

To determine whether tidal reinstatement and associated changes in vegetation distribution had any long-term impacts on migratory shorebird habitat use, I surveyed wetlands on Ash Island during low tides from 2004-06 (nine years after the removal of culverts), comparing my counts to low tide surveys before the reinstatement of tidal flushing (1994-95) and shortly after (1995-97) (Kingsford *et al.* 1998). I also investigated whether mangrove removal in three wetlands on Ash Island (2005-06) had any short-term impacts on shorebird habitat use by comparing high tide counts before (2004-05) and after (2006-07) the mangrove clearing.

7.3 Methods

7.3.1 Study site

The Hunter estuary borders the city of Newcastle, NSW (Fig. 7.1), which expanded following major industrial development of the estuary from 1891 onwards (Kingsford & Ferster Levy 1997; Williams *et al.* 2000) (for a review see Chapter 2). Many of the original islands and sandflats have been reclaimed during dredging of the Hunter River and the larger islands (Ash, Moscheto, Dempsey and Walsh) were connected and renamed Kooragang Island (Fig. 7.1) in 1967 (Moss 1983). Now a major feature of the lower estuary, the southern half of Kooragang Island is zoned for industrial use while its northern and western portions contain estuarine wetlands, partly reserved under the Hunter Wetlands National Park (Fig. 7.1) (DECC 2007). This area is internationally recognised under the Ramsar Convention and the East Asian-Australasian Shorebird Site Network. Up to 35 migratory shorebird species spend their non-breeding season (Sept-Apr) in the estuary, and two species have been recorded in numbers greater than 1% of their East Asian-Australasian Flyway (EAAF) populations and seven species in nationally significant numbers (> 1% of Australian populations) (see Chapter 2). The Kooragang dykes (Fig. 7.1) and Stockton sandspit provide major day-time roosting habitat for shorebirds, with many species foraging on nearby intertidal mudflats in the North Arm of the Hunter River and Fullerton Cove (Fig. 7.1) during low tides (see Chapters 4 and 5) (Geering 1995; Kingsford & Ferster Levy 1997).

The main study area is on the western end of Kooragang Island and is known as Ash Island. This area contains a complex of wetlands, bordered by a railway line and subdivided by roads, major power lines and a system of culverts and bridges that control inflows from the Hunter River through six creeks (Fig. 7.1). Ash Island supports about 10% of migratory shorebirds in the Hunter estuary during summer months (Sep-Apr), with 21 species recorded historically (1993-2007) (see Appendix C). Estuarine vegetation is dominated by the mangrove *Avicennia marina* and a mix of saltmarsh species, *Sacrocornia quinqueflora*, *Sporobolus virginicus, Suaeda australis* and *Triglochin striata*, surrounding shallow tidal pools (Outhred & Buckney 1983; Buckney 1987; Winning 1991). Ash Island's original vegetation included lowland rainforest species Swamp Oak *Casuarina glauca* and Blueberry Ash *Elaeocarpus obovatus* with areas of saltmarsh and mangrove (Buckney 1987), but the island was cleared for cattle grazing in the

1800s and artificial structures reduced inflows degrading estuarine vegetation (Kingsford & Ferster Levy 1997).

At the time of the KWRP's founding in 1993, Ash Island was dominated by pasture and degraded saltmarsh areas which were grazed by cattle (Fig. 7.2). The main task of the KWRP was to restore flows to this area and this was achieved through the removal of culverts on creeks 1 and 5 (Fig. 7.1) in 1995 (Streever *et al.* 1996). Studies that investigated vegetation responses to tidal manipulation in Area E (wetlands 12-19; Figs. 7.1; 7.2), the largest (124 ha) rehabilitation area on Ash Island, found that within one year (1995-1996) of culvert removal at Creek 5 saltmarsh had expanded upslope into pasture (Streever & Genders 1997). After three years (1998) of increased tidal flushing, saltmarsh had increased by 2.28 ha, mainly through expansion into upland pasture, while tidal pool/mudflat had increased by 5.58 ha and mangroves increased by 3.52 ha (MacDonald 2001) (Fig. 7.2). This increase in estuarine vegetation initially increased habitat for shorebirds in Area E but by 2004, follow-up photogrammetry indicated that there had only been a small increase in saltmarsh (0.99 ha) since 1998, and a substantial decrease in tidal pool/mudflat (6.86 ha) and a substantial increase in mangroves (7.68 ha) (Howe 2008) (Fig.7.2).

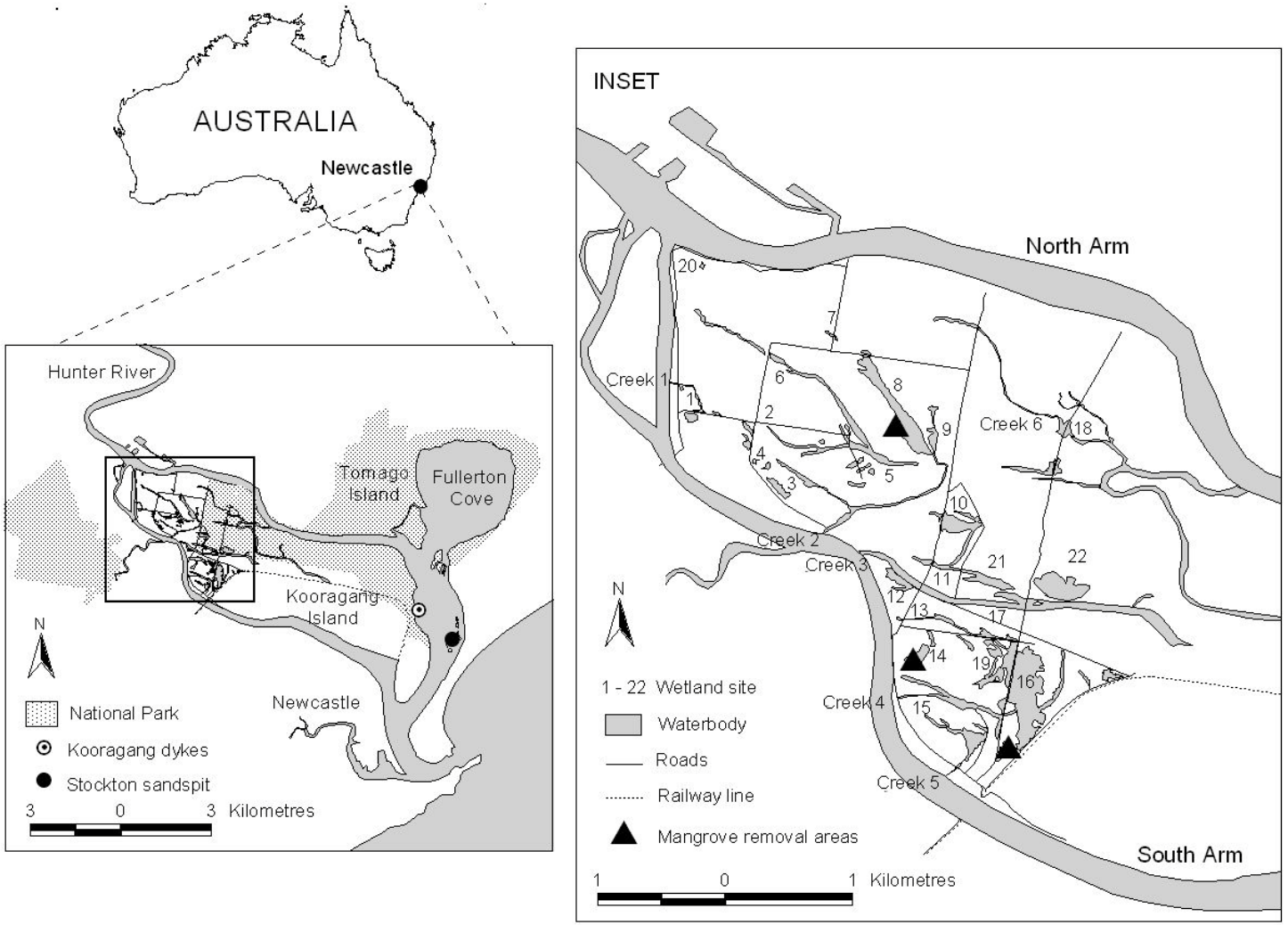
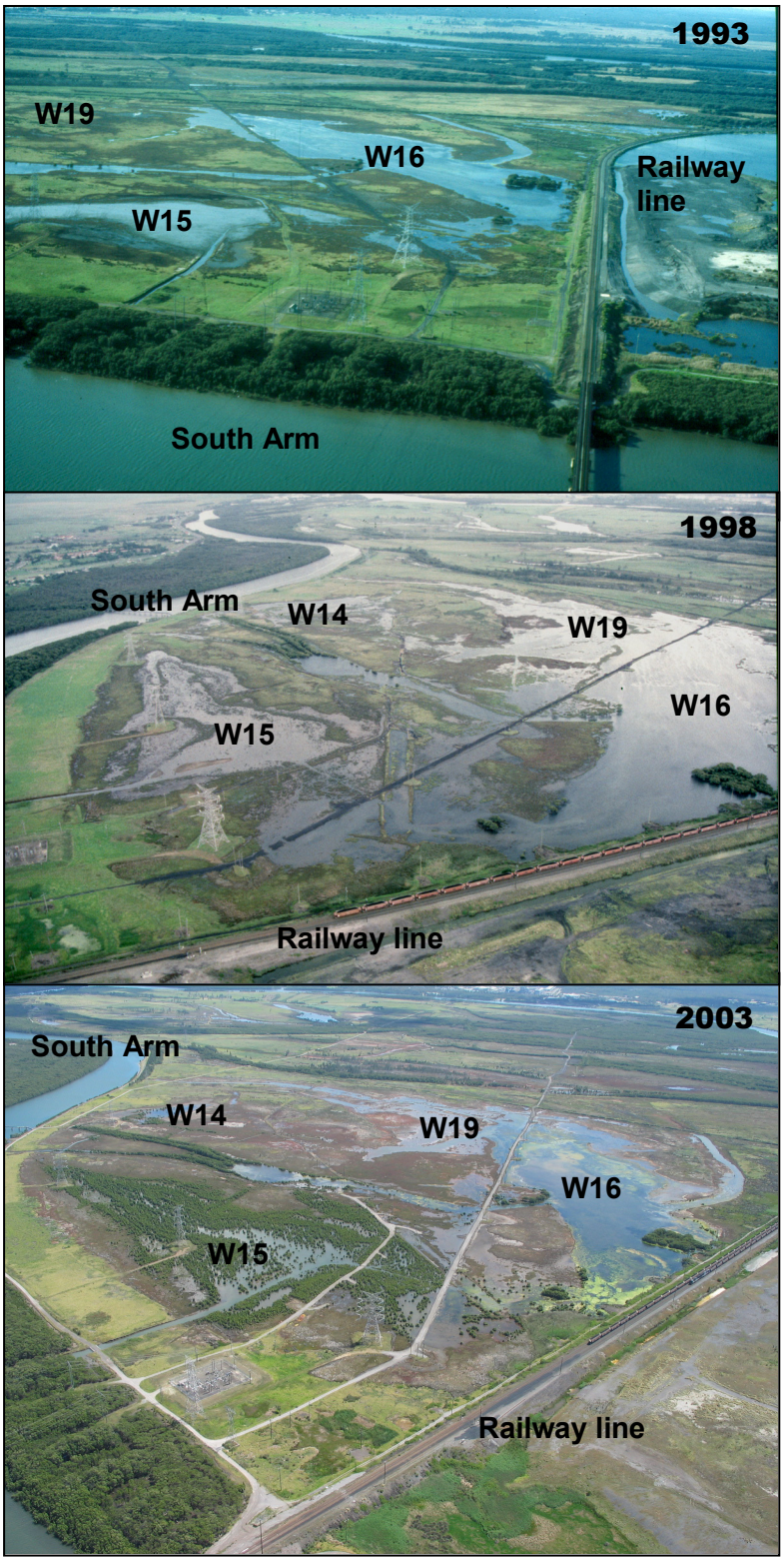


Figure 7.1 The Hunter estuary, including Kooragang Island, in south-eastern Australia. The main study site is part of the Kooragang Wetland Rehabilitation Project, Ash Island (inset), on the western half of Kooragang Island. This area was divided into wetland sites for low (sites 1-19) (1994-97; 2004-06) and high (sites 1-22) tide surveys (2004-05; 2006-07). Ash Island is flushed by six tidal creeks. Culverts were removed from creeks 1 and 5 between July and August 1995. Areas where mangrove was removed (2005-06) are indicated.



Credit: KWRP

Figure 7.2 Changes in vegetation distribution in Area E, Kooragang Island from 1993 (before culvert removal) to 1998 and 2003 (after culvert removal). Wetlands 14, 15, 16 and 19 are shown.

7.3.2 Shorebird surveys (1994-2007)

Migratory shorebirds were surveyed in the Hunter estuary between May 1994 and May 1997, which included repeated low tide surveys of wetlands on Ash Island (Kingsford *et al.* 1998) (see section 2.3.2). Ash Island was surveyed over three non-breeding seasons (Oct-Mar) of 1994-95, 1995-96 and 1996-97 (n = 53) at 19 wetland (sites 1-19 on Fig. 7.1), over three consecutive days (except in March 1997 when only two surveys were done). Ash Island tides lag the estuary mouth by about 1.4 hours (Kingsford *et al.* 1998) determining the order for surveys of tidal sites. The survey order for freshwater sites was randomised each time, unless they were adjacent to a tidal wetland, when they were surveyed together to limit disturbance. I counted migratory shorebird species on Ash Island, using the methodology of Kingsford *et al.* (1998), over two non-breeding seasons from 2004-06, except that no surveys were conducted in October 2004 (2004-05 (n = 15); 2005-06 (n = 18)) (see Appendix E1). Total numbers of shorebird species were counted using binoculars (Swarovski 8 x 30 mm) and a telescope (Swarovski 20 – 60X zoom). To investigate the effect of mangrove removal on roosting habitat availability, I conducted monthly high tide surveys of Ash Island over the 2004-05 (Nov – Mar (n = 15)) and 2006-07 (Dec – Mar (n = 12)) non-breeding seasons, during spring high tides (1.72 - 2.07 m) (see Appendix E1), when roosting habitat availability was limited in the estuary. The same methodology was used as for low tide surveys, except that three additional wetland sites (wetlands 20, 21 and 22) (Fig. 7.1) were added (see Appendix E2 for site descriptions).

7.3.3 Experimental design and statistical analysis

I used a two-way ANOVA with *post hoc* Tukey multi-comparison tests to determine whether low and high tide counts of migratory shorebirds differed among non-breeding seasons or survey months (Oct-Mar) on Ash Island. I also investigated whether high tide counts on Ash Island correlated with total numbers of migratory shorebirds in the entire Hunter estuary with a two-tailed Pearson's correlation (Quinn & Keough 2002). Counts for the Hunter estuary were from monthly surveys conducted by the Hunter Bird Observers Club (HBOC) (see Appendix C). Total counts were log transformed ($\log_{10}(x+1)$) to attain normal distributions and equality in the variances (Fowler *et al.* 1998). Differences in species assemblages during low tides were examined using the Bray-Curtis measure of similarity (Bray & Curtis 1957). Shorebird counts were averaged for each survey month and transformed with fourth root to control for multiple zeros and large values present in the data sets (Quinn & Keough 2002). The transformed

abundance data were subjected to non-metric multi-dimensional scaling (nMDS) (Clarke & Warwick 2001) to demonstrate patterns in shorebird species assemblages in tidal wetlands among surveys (1994-2006). I used a two-way crossed analysis of similarity (ANOSIM) design (Clarke & Warwick 2001) to test whether species assemblages differed in wetland sites among low tide surveys.

Before and After Control/Impact (BACI) designs were used to determine the effect of culvert and mangrove removal on migratory shorebird abundance on Ash Island. BACI sampling designs have been widely used in studies of environmental impacts on the abundance of populations, with sampling in impacted (treatment) and non-impacted (control) sites before and after a specific disturbance (Underwood 1992; Green 1993). The first experiment was designed so that low tide surveys were conducted before (1994-95) and after (1995-97; 2004-06) culverts were removed (Jul-Aug 1995) on creeks 1 and 5 to increase tidal flow in two experimental sites (wetlands 1 and 15 respectively), while tidal flow remained unchanged in two control sites (wetlands 3 and 14) (Fig. 7.1) (Kingsford *et al.* 1998). I investigated whether total numbers of migratory shorebirds differed between the experimental and control sites among the five non-breeding seasons using a Friedmans' test, a non-parametric equivalent to a one sample repeated measure design when counts are not normally distributed (Quinn & Keough 2002). A Kruskal Wallis test was used to determine if experimental and control sites differed overall (1994-2006).

In the second experiment, I compared migratory shorebird counts before (2004-05) and after (2006-07) mangrove trees (< 3 m, 5 – 10 years old) and seedlings were removed in three treatment sites (wetlands 8, 14 and 16 (south)) (Figs.7.1; 7.3) (8.5 ha in total) from August 2005 to August 2006. These counts were compared to three control sites (wetlands 5, 15 and 19 (south)) (Figs. 7.1; 7.4), which contained similar sized mangrove trees that were not removed, and two reference sites, which were open saltmarsh (wetlands 16 (north) and 19 (north)) (Fig. 7.1). My hypothesis was that migratory shorebird abundance would increase in wetlands where mangrove was removed compared to the control sites, and instead approach the reference sites (Grayson *et al.* 1999) which were already utilised by migratory shorebirds. I used a two factor logistic regression analysis (Quinn & Keough 2002) to investigate whether the presence of migratory shorebirds increased in treatment sites after mangrove was removed compared to the control and reference sites. Univariate analyses were done in SPSS (2005) and multivariate analyses in PRIMER (2002). Means and standard errors are presented throughout.

Credit: A. Howe and J. Spencer.

Figure 7.3 Mangrove removal from the three experimental sites: upper row before mangrove removal (May-Aug 2005) and lower row after removal (Dec 2005-Aug 2006) (left to right: wetlands 16 (south) (W16S), wetland 8 (W8) and wetland 14 (W14) (see Fig. 7.1 for locations).

Credit: A. Howe and J. Spencer.

Figure 7.4 Wetland control sites for mangrove removal experiment (2004-07): upper row (2004-05) and lower row (2006-07) (left to right: wetland 15, wetland 5 and wetland 19 (south)) (see Fig. 7.1 for locations).

7.4 Results

7.4.1 Low tide distribution and effects of tidal reinstatement (1994-2006)

Overall, total numbers of migratory shorebirds were highly variable among low tide surveys, differing among non-breeding seasons ($F_{4, 57} = 25.0$, $p < 0.001$) and survey months ($F_{5, 57} = 7.7$, $p < 0.001$) (Fig. 7.5). Total counts were highest in the 1994-95 (207.9 ± 43.2 birds) and 2004-05 (416.0 ± 150.2 birds) non-breeding seasons and lowest in the early stage of each season (Oct-Nov) but there was also a significant interaction between season and month ($F_{19, 57} = 10.7$, $p < 0.001$), e.g., exceptionally high counts (789 - 2,128 birds) recorded during surveys in March 2005 were due to large numbers of Sharp-tailed Sandpipers *Calidris acuminata* (1,120.3 ± 279.2 birds) using the study site but large flocks (200 - 500 birds) were also observed in December 1994 and January 2005 (Figs. 7.5; 7.6). Total numbers of species were similar among non-breeding seasons (nine species in total: 1994-2006), but only four species were recorded every season (Common Greenshank *Tringa nebularia*, Eastern Curlew *Numenius madagascariensis*, Marsh Sandpiper *Tringa stagnatilis* and Sharp-tailed Sandpiper) (Fig. 7.6; Appendix E3). Most species avoided freshwater sites, only Latham's Snipe *Gallinago hardwickii* preferred this habitat over the tidal sites (Appendix E3).

Shorebird distribution changed over the survey period (Figs. 7.7; 7.8). Total shorebird counts were low for many of the wetland sites, but fewer sites were used during low tide surveys in 2004-06 (13 of 19 sites) compared to surveys in the 1990s (14 sites in 1994-95; 16 sites in 1995-97) (Fig. 7.7). Overall, wetland 16 supported the most shorebirds, but there was greater use of wetland 19 in 2004-06 (Fig. 7.7). Although there appeared to be some clustering of the tidal wetlands in the survey periods after multidimensional scaling (wetlands 8, 10, 15, 18 and 19 in 1994-95; wetlands 5, 8, 9, 15, 16, 19 in 1995-97; and wetlands 16 and 19 in 2004-06:) (Fig. 7.8), no significant differences in species assemblages were detected among sites (*global* $R = 0.12$, $p = 0.235$) across the non-breeding seasons (*global* $R = 0.07$, $p = 0.655$).

Total numbers of migratory shorebirds did not differ significantly between experimental (wetlands 1 and 15) and control sites (wetlands 3 and 14) ($X^2 = 1.9$, df = 1, $p = 0.167$) (1994 - 2006), but there were significant declines in total numbers of shorebirds in all sites after culvert removal ($X^2 = 92.6$, df = 9, $p < 0.001$) (Fig. 7.9). Despite high variability among survey months, all four sites were used regularly by migratory shorebirds during 1994-95 (Fig. 7.9), with highest

species diversity on wetland 15 (seven species), but only five species were recorded in surveys in 1995-97 and only one species (Eastern Curlew) in 2004-06 surveys. Similarly, shorebird use declined in the nearby control site (wetland 14), from four species in 1994-95 and 1995-97 surveys to two species (Sharp-tailed Sandpiper and Eastern Curlew) in 2004-06 (Fig. 7.7a). In 1994-95, five migratory shorebird species were recorded in wetland 3 (control) and four species in wetland 1 (experimental), including flocks of Pacific Golden Plovers *Pluvialis fulva* (45 - 61 birds), but no migratory shorebirds were observed in either wetland during low tide surveys in 2004-06 (Figs 7.7; 7.9).

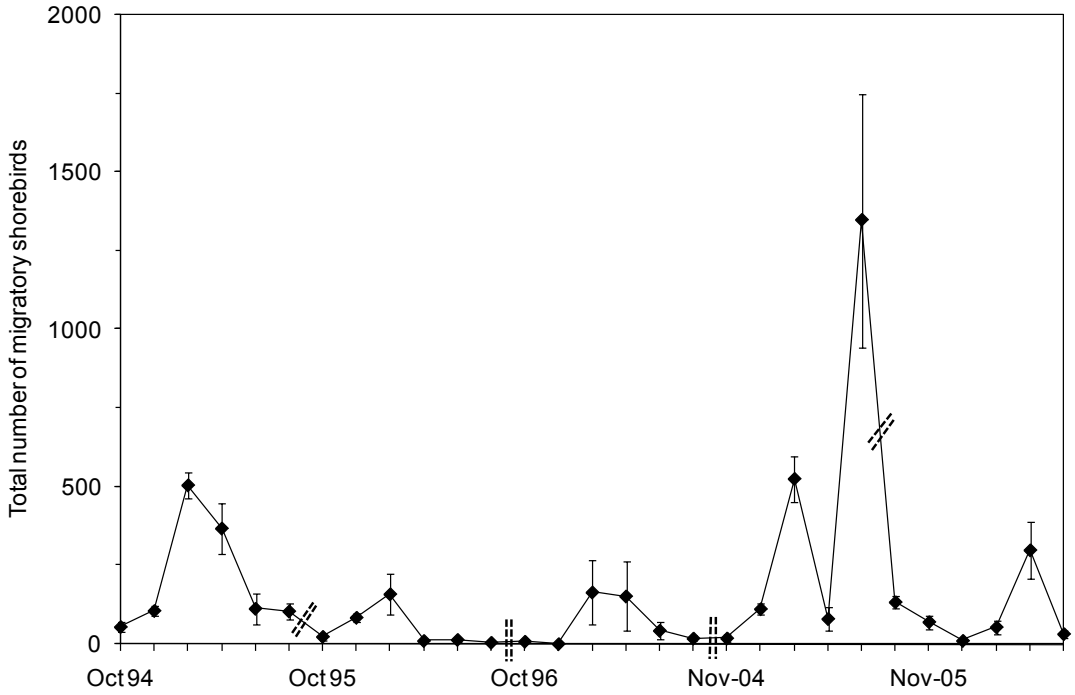


Figure 7.5 Mean (± S.E.) total number of migratory shorebirds observed on Ash Island during low tide summer surveys (Oct 1994-Mar 1997; Nov 2004-Mar 2006 (n = 86)). Culverts were removed in Jul-Aug 1995, after the first non-breeding season (Oct 1994-Mar 1995) (dashed lines indicate breaks between non-breeding seasons).

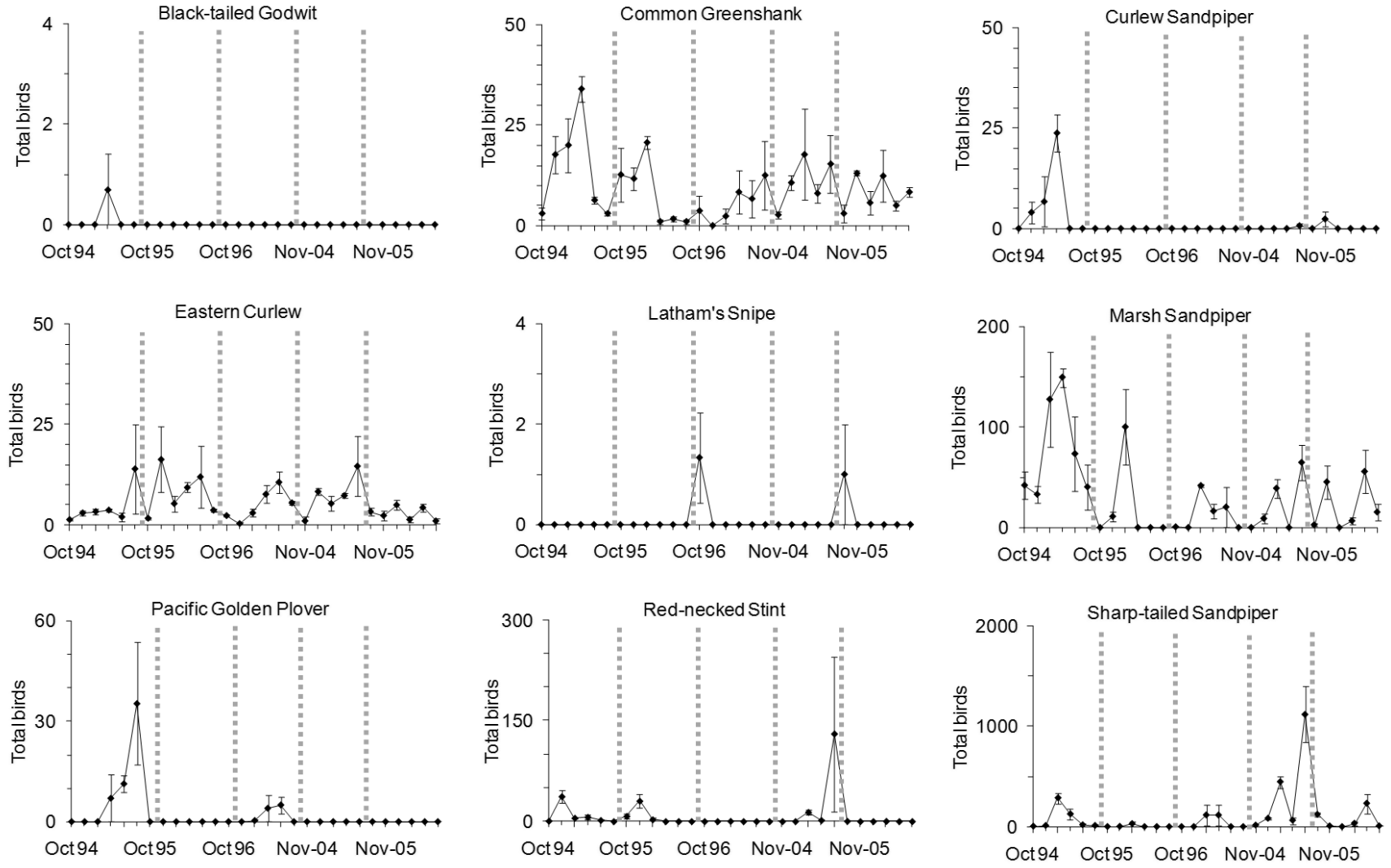


Figure 7.6 Mean (± S.E.) numbers of nine migratory shorebird species observed during low tide surveys of Ash Island (Oct 1994-Mar 1997; Nov 2004-Mar 2006 (n = 86)) (dotted lines indicate breaks between non-breeding seasons).

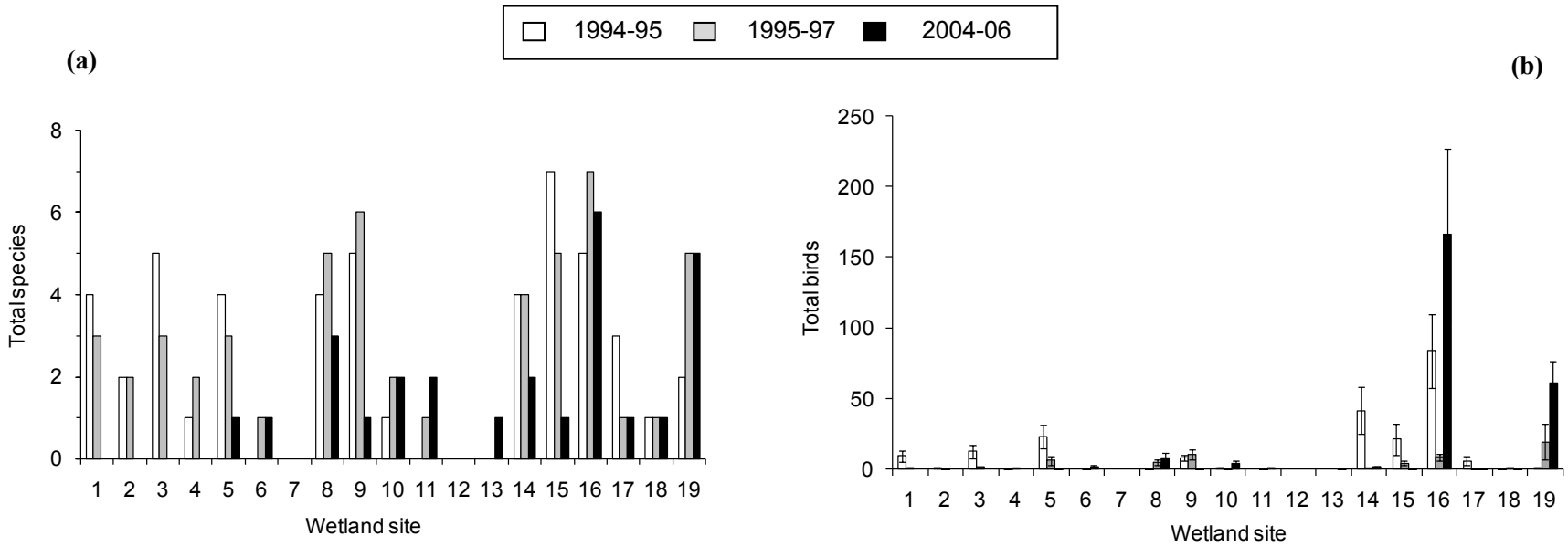


Figure 7.7 Total number of migratory shorebird (a) species and (b) mean (± S.E.) counts observed in 19 wetlands on Ash Island during low tides before (1994-95) and after (1995-97; 2004-06) tidal reinstatement (see Figure 7.1 for locations and Appendix E2 for site descriptions).

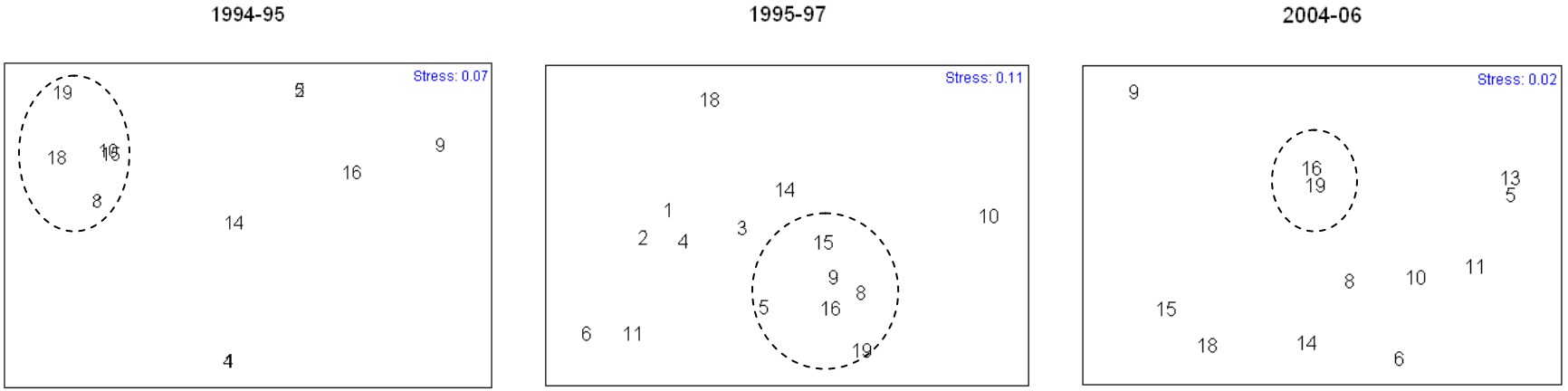


Figure 7.8 Non-metric multi-dimensional scaling (nMDS) plots of wetlands on Ash Island according to the mean abundance of nine migratory shorebird species observed during low tide surveys before (1994-95) and after (1995-97; 2004-06) tidal reinstatement (see Fig. 7.1 for locations).

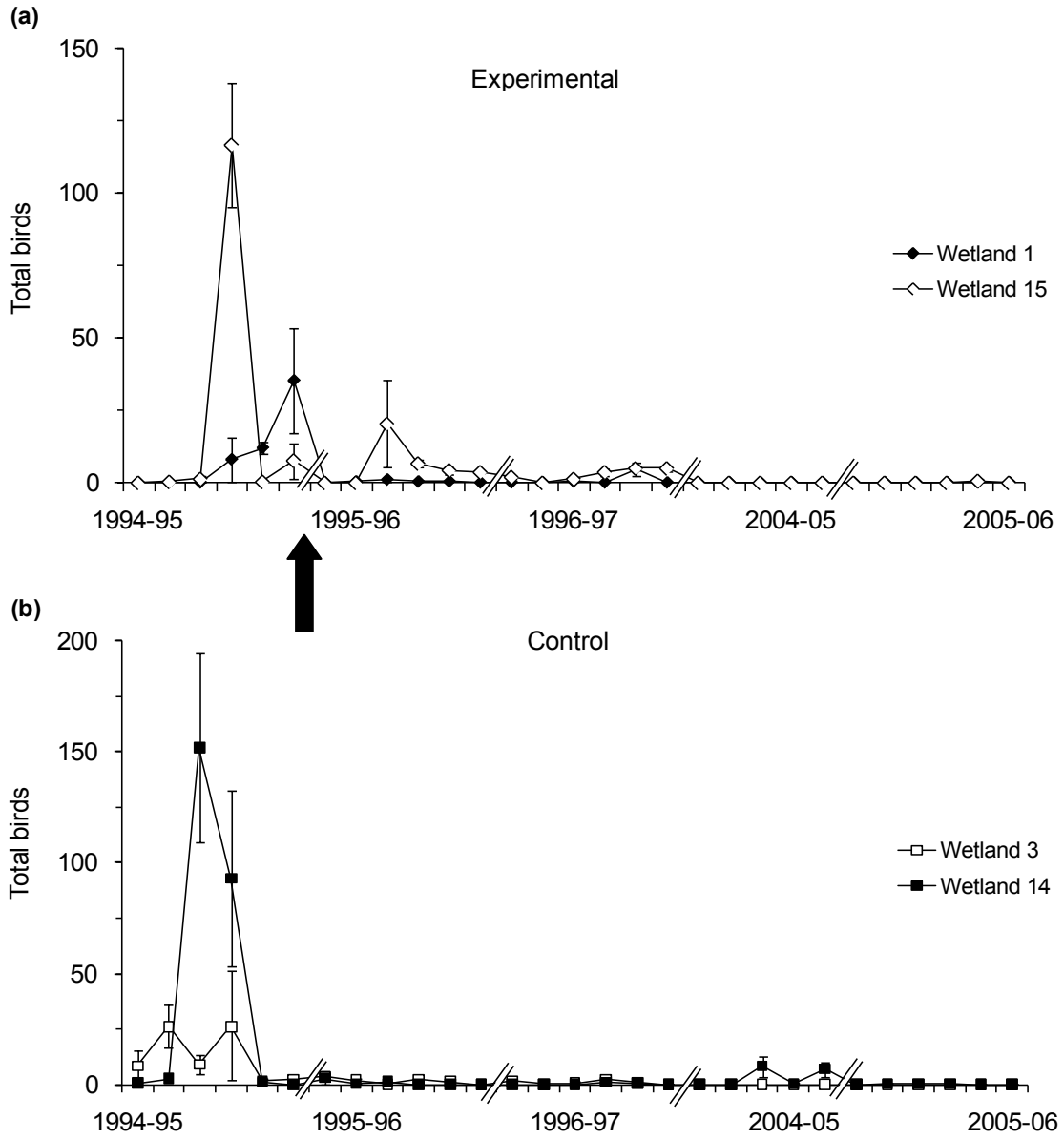


Figure 7.9 Mean (± S.E.) total number of migratory shorebirds recorded in (a) experimental (wetlands 1 and 15) and (b) control (wetlands 3 and 14) sites during low tide surveys of Ash Island before (Oct 1994-Mar 1995, n = 18) and in two periods (Oct 1995-Mar 1997, n = 35; Nov 2004-Mar 2006, n = 33) after tidal reinstatement (arrow).

7.4.2 High tide distribution and effects of mangrove removal (2004-2007)

There were significant differences in total numbers of migratory shorebirds during high tide surveys of Ash Island (year: $F_{1,18}$ = 11.0.0, p = 0.004; month:$F_{4, 18}$ = 4.0, p = 0.017; survey x month $F_{3, 18}$ = 3.6, p = 0.035). The largest counts were observed in the 2004-05 non-breeding season, peaking in January to March 2005, but counts on Ash Island did not correlate with total numbers observed in the whole of the Hunter estuary (r = 0.10, n = 9, p = 0.808) (Fig. 7.10). Migratory shorebirds distribution was similar between high and low tide surveys (2004-07) (Figs. 7.7; 7.11): wetlands 16 and 19 supported the most migratory shorebirds during day time high tides, while few shorebirds were observed on the remaining sites and no shorebirds were observed on wetlands 1, 2, 3, 4, 12 or 13 or freshwater sites (Fig. 7.11; Appendix E2).

Twelve species were recorded during high tide surveys (2004-07) (Fig. 7.12). Large numbers (300 - 600) of Sharp-tailed Sandpipers and Marsh Sandpipers (50 – 100 birds) were recorded in wetlands 16 and 19. Flocks of Eastern Curlew (10 - 30 birds) were also common in wetland 19. Single Black-tailed Godwits *Limosa limosa* were observed in both survey years and Double-banded Plovers *Charadrius bicinctus* and a single Ruff *Calidris pugnax* in March 2007 (Fig. 7.12). Several species also roosted on the riverbank on the North Arm of the Hunter River (near wetland 20) (Fig. 7.1), including Common Sandpipers *Actitis hypoleucos*, Whimbrels *Numenius phaeopus*, Common Greenshanks and small flocks of Pacific Golden Plovers (10 - 30 birds). Flooded saltmarsh in wetland 21 and wetland 22 (Fig. 7.1) provided habitat for Sharp-tailed Sandpipers, Common Greenshanks and Marsh Sandpipers.

After mangrove removal, treatment sites did not differ significantly from the control sites but both differed from the reference sites (Table 7.1; Fig. 7.13). Although there were fewer shorebirds in the 2006-07 season, the reference sites (wetlands 16 (north) and 19 (north)) supported a high proportion of shorebirds observed on Ash Island in both non-breeding seasons (260.6 ± 64.1 birds (2004-05); 103.7 ± 27.3 birds (2006-07)). Numbers of migratory shorebirds were low in the control sites (wetlands 5, 15 and 19 (south)) in both non-breeding seasons (0.1 ± 0.1 birds (2004-05); 0.8 ± 0.3 birds (2006-07)) (Fig. 7.13). Small numbers of the most common species (Common Greenshank, Eastern Curlew, Marsh Sandpiper and Sharp-tailed Sandpiper) were observed in the treatment areas before (8.2 ± 4.6 birds) and after (14.3 ± 4.6 birds) mangrove removal (Fig. 7.13).

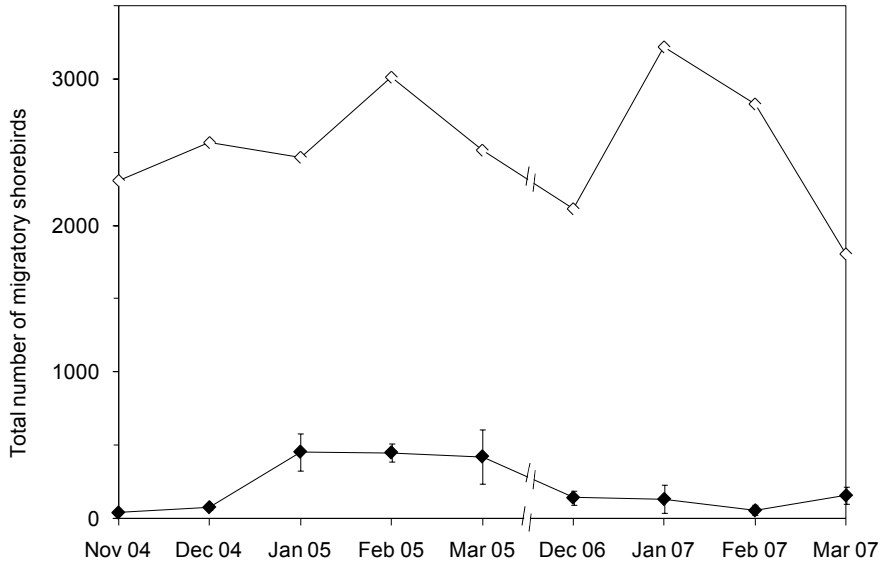


Figure 7.10 Mean (± S.E.) total number of migratory shorebirds observed on Ash Island (◆) (n = 27) during high tide surveys compared to total monthly counts (n = 9) for the entire Hunter estuary (◇) (including Ash Island) (Nov 2004-Mar 2005; Dec 2006-Mar 2007).

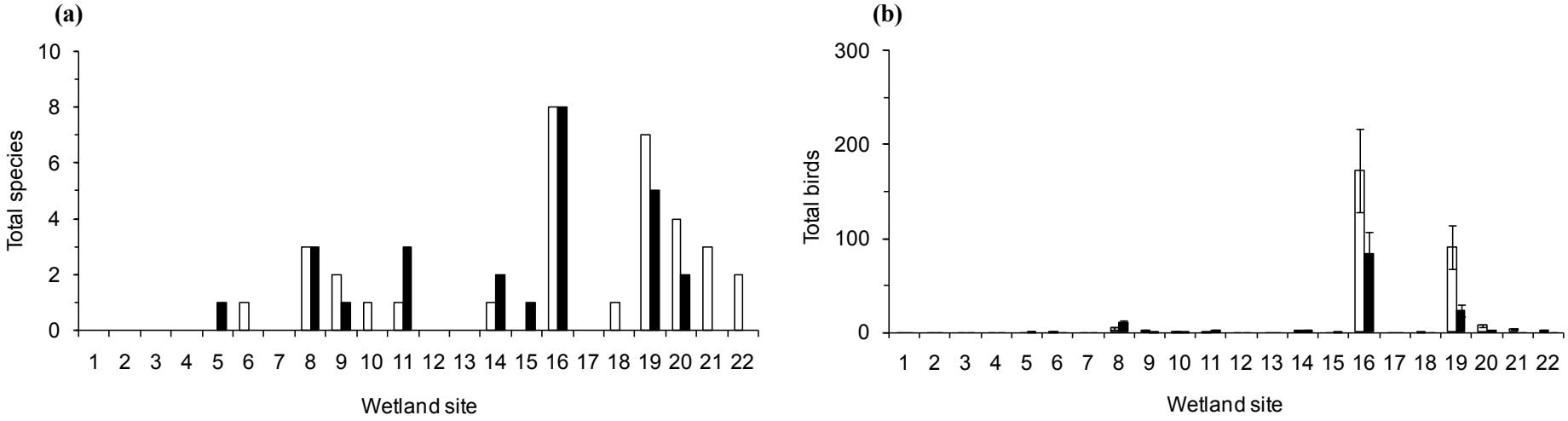


Figure 7.11 Total number of migratory shorebird (a) species and (b) mean (± S.E.) counts observed during high tides surveys of 22 wetlands on Ash Island in summer months in 2004-05 (open bars) and 2006-07 (filled bars) (see Figure 7.1 for locations and Appendix E2 for site descriptions).

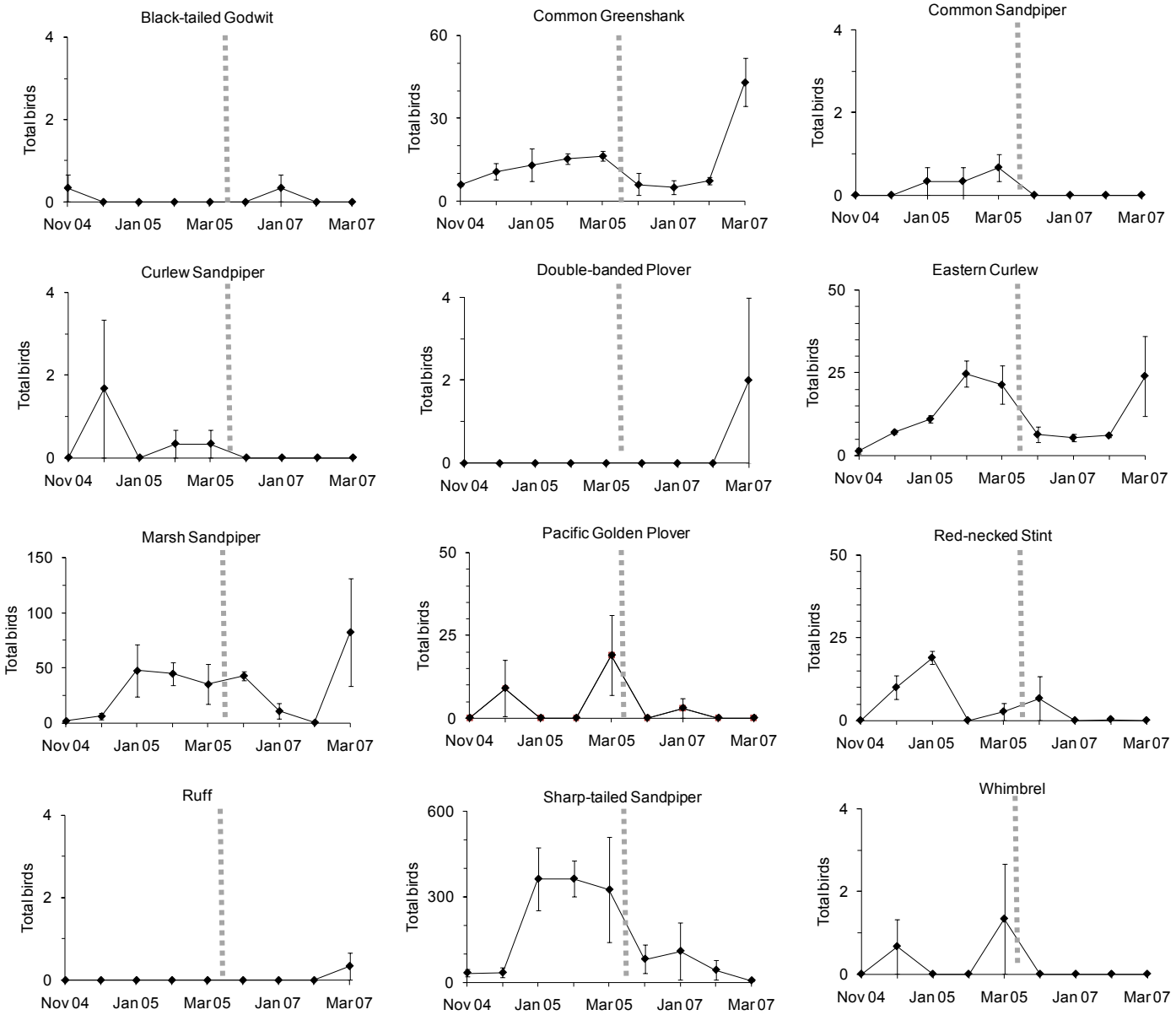


Figure 7.12 Mean (± S.E.) numbers of twelve migratory shorebird species observed during high tide surveys of Ash Island (Nov 2004 – Mar 2005; Dec 2006 - Mar 2007) (dotted lines indicate breaks between survey periods).

Table 7.1 Results of binary logistic regression of migratory shorebirds observed during high tide surveys of wetlands on Ash Island in relation to site type (treatment, control and reference) and survey year (2004-05; 2006-07).

Variable	B	S.E.	*Wald*	df	*p*
Site type			9.9	2	**0.007**
Site type (1)	-3.8	2.2	2.5	1	0.114
Site type (2)	6.1	2.4	6.4	1	**0.011**
Year	1.5	0.5	9.3	1	**0.002**
Site type * Year			1.8	2	0.409
Site type (1) by Year	0.7	1.2	0.4	1	0.555
Site type (2) by Year	-1.7	1.5	1.3	1	0.264
Constant	-2.5	0.8	10.7	1	0.001

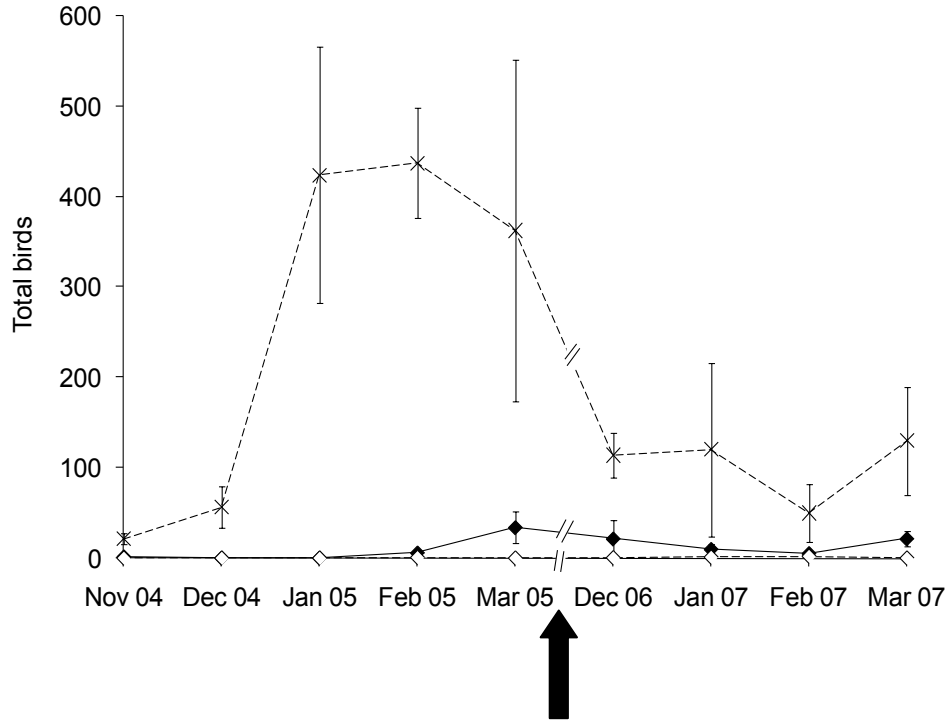


Figure 7.13 Mean (± S.E.) numbers of migratory shorebirds recorded in treatment (◆), control (◇) and reference (x) sites on Ash Island during high tide surveys (2004-05; 2006-07) (see Methods for locations). Mangrove was removed (arrow) from the treatment sites from Aug 2005 - Aug 2006 (see Fig. 7.3).

7.5 Discussion

Port development has altered the Hunter River's tidal regime and the distribution of estuarine vegetation on Ash Island (Buckney 1987; MacDonald 2001), with mangroves increasing by 74% from 1954 to 1994 (a rate of 0.5 ha yr^{-1}) (Williams *et al.* 2000). The migration of mangrove into saltmarsh is a widespread phenomenon in south-eastern Australia (Saintilan & Wilton 2001; Wilton 2002) and has been attributed to: altered tidal regimes; urbanisation; increased rainfall, nutrients, and sedimentation; and sea level rise and subsidence of intertidal flats (Saintilan & Williams 1999).

Rehabilitating intertidal habitats can be problematic because they are often complex systems (Moy & Levin 1991; Zedler 2004) and it often difficult to maintain field experiments over long time frames. Tidal manipulation indirectly reduced habitat availability for shorebirds in Area E by promoting mangrove colonisation of former shorebird habitat (Howe 2008). In this study, wetlands with large stands of mangroves were rarely used by shorebirds during low or high tide periods. This pattern was widespread with mangrove increasing in experimental and control sites, although at least one control site (wetland 14) was compromised by increased hydrological connectivity with a neighbouring catchment (wetland 15) after tidal reinstatement (MacDonald 2001; Howe 2008).

On Ash Island the largest concentrations of shorebirds were in open areas of saltmarsh that surrounded shallow tidal pools with few mangroves (wetlands 16 and 19) (Figs. 7.7; 7.11), and these were the only sites where water levels and mangroves were artificially controlled by culverts. Prior to culvert removal in 1993, approximately 31.5 ha of saltmarsh, mudflat and shallow tidal pool were available to shorebirds in Area E. Following culvert removal, shorebird habitat availability was reduced by 17% (5.4 ha) due to the expansion of mangroves in Area E (Howe *et al.* in press). During 2004-06 there was a reduction in both shorebird diversity and total abundance, particularly in wetlands 14 and 15 (Fig. 7.7), where extensive mangrove encroachment had occurred (Fig. 7.2). In the 1994-1997 survey period, wetland 15 supported seven species of migratory shorebirds, but only one species (Eastern Curlew) was recorded in the 2004-2006 surveys. Wetland 14 also provided low tide habitat for four species during 1994-1997, but only small numbers of Eastern Curlew and Sharp-tailed Sandpiper (1-15 birds/ flock) were recorded during the 2004-2006 surveys (Fig. 7.7). In contrast, numbers of shorebirds

remained high in wetlands 16 and 19 where open areas of saltmarsh and tidal pools remained accessible.

The effects of mangrove removal in Area E were minimal, but there was limited opportunity to detect change during this study because of significant variation in total numbers of migratory shorebirds within and between non-breeding seasons (Fig. 7.13). My analysis of low tide counts of shorebirds on Ash Island from 1994-2006 illustrated the importance of long term data sets for the detection of changes in shorebird abundance and habitat use in response to wetland manipulation. Further surveys are required to determine the effectiveness of mangrove removal for restoring shorebird habitat on Ash Island, preferably using the same methodology to permit comparisons across years and sites.

High levels of variability in shorebird numbers in Ash Island and the Hunter estuary can be explained by local, regional, and/or flyway effects on migratory shorebird populations (see Chapter 2) (Gosbell & Clemens 2006; Nebel *et al.* 2008). Wetland availability on Ash Island is driven by fortnightly spring tidal cycles, local rainfall and evaporation (Howe 2008), which affect the distribution of shorebirds as they generally prefer habitats with shallow water or wet substrates not deeper than their thigh (Dann 1987; Skagen & Knopf 1994; Parsons 2002) (see Table 4.1). During surveys in 2006-07 many sites were dry or water levels were low, which coincided with a period of low rainfall (BOM 2007) and low numbers of shorebirds (Fig. 7.10). For the most part, large fluctuations in total numbers of shorebirds on Ash Island were caused by Sharp-tailed Sandpipers and Marsh Sandpipers. Sharp-tailed Sandpipers normally peak in south-eastern Australia between late January and February each year (Lane 1987), but both species will move between coastal and inland wetlands in response to wetland availability and prevailing climatic conditions (Thomas 1970; Higgins & Davies 1996). Sharp-tailed Sandpipers may have benefited from increases in saltmarsh availability on Ash Island as it is an important feeding habitat for this species (see Chapter 6).

A further 19 species have been observed on Ash Island at different times (1993 - 2007) (see Appendix C) contributing to the variation in abundance of migratory shorebirds. These have included: the threatened Black-tailed Godwit (*NSW Threatened Species Conservation Act 1995*); vagrant Lesser Yellowlegs *Tringa flavipes*; large flocks of Red Knots *Calidris canutus* and Red-

necked Stints *Calidris ruficollis* which stop en route to non-breeding grounds further south of the Hunter estuary; and roosting Pacific Golden Plovers and Eastern Curlews which normally spend high tide periods at day roosts in the Hunter River.

Migratory shorebird species are faithful to a wetland complex during their non-breeding season (Dann 1981a; Driscoll 1995; Rehfisch *et al.* 1996) but will move to habitats opportunistically as they become available (Skagen & Knopf 1994; Atkinson *et al.* 2004). Before mangroves were removed on Ash Island, shorebirds only used treatment sites when high numbers were observed in surrounding wetlands (e.g., March 2005), which perhaps caused some 'overspill' of shorebirds into less favourable habitats. Some species, such as the Sharp-tailed Sandpiper, may also have a higher tolerance for feeding near vegetation when prey densities are high (Taylor 2003). Migratory shorebirds were observed in the cleared sites in every survey month after mangroves were removed, despite total numbers of shorebirds being low on Ash Island in the 2006-07 non-breeding season (Fig. 7.13). In contrast, when mangroves were removed from Stockton sandspit in the North Arm of the Hunter River, there was an immediate uptake of the cleared habitat by hundreds of migratory shorebirds (Herbert 2007), but this site is close to the Kooragang dykes, a well established day roost, which supports large numbers of shorebirds during high tide periods (see Chapter 3) (Geering 1995).

Many studies have highlighted the importance of a mosaic of wetland sites for conserving migratory shorebird populations (Haig *et al.* 1998; Neckles *et al.* 2002). In Australia limited emphasis is placed on the importance of ephemeral habitats for shorebird populations in management plans. In the Hunter estuary the most important wetlands for shorebirds on Ash Island are outside the boundaries of the Hunter Wetlands National Park (see Fig. 7.1). As a signatory to international migratory bird agreements and conservation conventions, the Australian government has obligations to protect migratory shorebird roosting and feeding habitat. Despite these saltmarshes supporting internationally significant numbers of Sharp-tailed Sandpipers (maximum counts exceed 1% of the East Asian-Australasian flyway population) and bordering a Ramsar-listed wetland, major threats to wetlands on Ash Island exist. Under NSW State Environmental Planning Policies (SEPP 74 and SEPP (*Major Projects*)) Ash Island could be developed for a major transport and infrastructure corridor to facilitate the expansion of Newcastle's ship loading capacity (NPC 2009). Shorebird habitat on Ash Island is also under

threat from further mangrove encroachment, as a result of eustatic sea level rise and the consequences of major dredging in the Hunter River (Howe 2008).

If restoring saltmarshes for shorebirds is the main objective tidal connectivity is important, but water levels should be managed to maintain shallow ponds with short sparse vegetation, while preventing the establishment of mangroves. This can be done at a local scale by using hydraulic controls to prevent mangrove propagules establishing (Clarke & Myerscough 1993), preferable to the manual removal of mangroves which may only be effective in the short-term (5 - 10 years) (Howe 2008); but estuary and catchment-scale approaches would have long-term benefits because the underlying causes of changes in mangrove/saltmarsh dynamics operate at much larger scales. There is a need for stronger control guidelines for new developments adjoining coastal saltmarsh, and/or the provision of buffer zones to allow the landward migration of these habitats. Dredging works in the Hunter River and global sea level rise will increase mean estuary water levels allowing greater migration of mangrove upslope (Howe 2008) and causing further reductions in migratory shorebird habitat in the Hunter estuary.

CHAPTER 8: GENERAL DISCUSSION

8.1 Overview

Like most animals, shorebirds need to feed, rest, find a mate and reproduce successfully while avoiding predation. As most migratory shorebirds breed thousands of kilometres from their non-breeding grounds, their survival is dependent on the availability of good quality roosting and foraging habitats along their entire flyway. In this thesis I focused on the Hunter estuary, a small part of the East Asian-Australasian Flyway (EAAF), but a non-breeding site where large-scale habitat loss has occurred and remaining shorebird habitats are under threat or degraded. I investigated the current status of migratory shorebird populations in this site, how the most common species used remaining estuarine habitats, and the effectiveness of rehabilitation efforts.

In this final chapter I discuss my findings in the context of migratory shorebird ecology in south-eastern Australia and links developed with the concurrent ecohydraulics project (Howe 2008). I also discuss the implications of my project for shorebird habitat rehabilitation and management in the Hunter estuary and provide recommendations for management and further research.

8.1.1 Population status and impacts of habitat loss

The Hunter estuary now supports about 3,000 migratory shorebirds (up to 35 species) in summer months (Sept-Apr). It is regionally important, supporting seven vulnerable species in NSW (*Threatened Species Conservation Act (TSC) 1995*) and large flocks of staging species before they reach non-breeding grounds further south of the Hunter estuary. It is also nationally important, supporting more than 1% of Australian populations of seven species, internationally significant numbers of two species (> 1% flyway population) and two globally threatened species (IUCN Red List) (see Tables 1.1, 2.3 and 2.6).

Historical counts of migratory shorebirds in the Hunter estuary only dated back to 1965, limiting my ability to assess whether habitat lost to port development in the preceding years had caused declines in total numbers of migratory shorebirds. Maximum counts from 1965-81 indicated that the Hunter estuary's migratory shorebird population was between 10,000 and 12,000 birds within this period, but maximum counts fell by about 34% (to 8,600 birds) and mean summer counts by 42% in the following 26 years (Table 2.3; Fig. 2.4). There was evidence for decline in at least four migratory shorebird species, with most severe declines seen in Curlew Sandpipers *Calidris*

ferruginea (> 80% since 2000) (Fig. 2.5; Table 2.5), but high inter-annual variability among counts made it difficult to detect change, even for relatively common species (see section 2.4.3).

We are perhaps only beginning to appreciate the cumulative effects of habitat loss and degradation on migratory shorebird populations. Declines in the Hunter estuary reflected similar trends in other non-breeding sites in south-eastern Australia, including Corner Inlet and the Shoalhaven estuary (Fig. 2.3), but it is difficult to determine whether habitat loss at the local scale or other parts of the EAAF have had most impact on shorebird populations in south-eastern Australia. Given that there has also been a 61% decline in shorebirds remaining in the Hunter estuary over winter months (1981-2007) (see section 2.4.1), typically first-year birds, it may be that declines in some species are a result of reduced recruitment and survival. There is already strong evidence that habitat alteration in key stop-over sites can directly impact shorebird populations. In the Netherlands, cockle *Cerastoderma edule* harvesting in the Wadden Sea reduced adult Red Knot *Calidris canutus islandica* survival causing a 25% decline in total numbers in north-western Europe (Van Gils *et al.* 2006) and horse-shoe crab harvesting in Delaware Bay, in the eastern United States, reduced fuelling rates for Red Knots *C. c. rufa* decreasing adult survivorship and the recruitment of second year birds by 37% and 47% respectively (Baker *et al.* 2004). Similarly, the reclamation and development of stop-over sites in the EAAF, in the Yellow Sea in particular, has destroyed intertidal systems and dependent shorebird populations have collapsed (Moores 2006; Moores *et al.* 2008).

8.1.2 Roost availability

Roosting habitat in the Hunter estuary took many forms including: artificial structures; sandbanks; staging intertidal mudflats; and saltmarshes. Shorebirds also used intertidal mudflats in the North Arm of the Hunter River as staging sites during flood and ebb tides but the use of mudflats varied with small species using these sites to maximise their feeding time while larger species preferred to carry out maintenance activities and rest (see section 4.4). Eastern Curlews *Numenius madagascariensis* roosted in different habitats during day and night high tides, preferring open sites near the main feeding habitats during the day and inundated saltmarsh at night. Eastern Curlews rested at the staging and major day and night roosts, but their need to sleep was greater at night (see section 3.4.2). Roost characteristics were important for shorebirds, with most species avoiding tall vegetation including mangroves. In roost design, a buffer of at least 10 - 40 m from tall vegetation (>5 m) is recommended (Lawler 1996) but the provision of shallow water is also

important (Rogers *et al.* 2006a). Shallow water was attractive to Eastern Curlews at day and night roosts, likely providing a mechanism for cooling on warm days and for detecting predators at night.

Howe (2008) demonstrated that high tide roosting area was extremely limited in the Hunter estuary, particularly during spring high tides, but for Eastern Curlews this was also restricted during periods of high disturbance and immediately prior to their migration (Feb-Mar) (see sections 3.4.1 and 3.4.3). The main day roosts experienced high rates of disturbance but mostly from birds of prey. Human-induced disturbances were less predictable than natural disturbances because they could take many forms, often causing prolonged periods of flight (up to 20 minutes in a single high tide period). Although I did not measure the effects of disturbance on the fitness of individual birds, similar rates of disturbance (0.8 - 1.7 per hour) have reduced the survival of shorebird species in the northern hemisphere, especially when food supplies were limiting (Durell *et al.* 2005; Goss-Custard *et al.* 2006).

8.1.3 Use of intertidal mudflats

The proximity of high tide roosting habitat is a key factor in determining shorebird feeding distribution (Lawler 1996; Dias *et al.* 2006; Rogers *et al.* 2006a) but travelling distances can vary according to the size of a shorebird. In the Hunter estuary, most small shorebirds foraged in mudflats along the foreshore of the lower estuary near the main day roosts, while larger Bar-tailed Godwits *Limosa lapponica*, Black-tailed Godwits *Limosa limosa*, Eastern Curlews and Whimbrels *Numenius phaeopus* often travelled further (3.5 - 7 km) to feed on intertidal mudflats in Fullerton Cove (Fig. 8.1). Artificial mudflats impounded by the Kooragang dykes (Fig. 8.1) were often the preferred feeding habitat of small shorebird species but these sites increased in importance for all shorebirds during neap low tides and one to three hours before high tide when the availability of intertidal mudflats was limited in the rest of the Hunter estuary (see Chapters 4 and 5). At times the Kooragang dyke impoundments could support more than 50% of migratory shorebirds in the Hunter estuary, including internationally significant numbers of Eastern Curlews and nationally significant numbers of Pacific Golden Plovers *Pluvialis fulva* (Table 4.2).

The availability of foraging habitat varied within and between daily tidal cycles, but shorebirds and benthic invertebrates were not evenly distributed across intertidal mudflats. Although benthic invertebrate species diversity was low, invertebrate density could be high (Fig. 5.5), which is

typical of temperate Australian estuaries (Jones *et al.* 1986; Morrisey *et al.* 1992; Winberg *et al.* 2006), however, this one-off sampling limited my ability to link differences in shorebird distributions to benthic invertebrate assemblages and abiotic factors. There were some evidence of relationships between shorebird and prey distributions (Fig. 5.7) and larger numbers of Bar-tailed Godwits, with high capture rates, were seen in Fullerton Cove where benthic invertebrates could be abundant.

Further studies of relationships between shorebirds and their prey are needed in the Hunter estuary. These studies should include detailed diet analysis of key shorebird species and further studies of benthic invertebrate assemblages (see recommendations in section 8.3). Such studies will help determine the vulnerability of shorebirds to changes in food supplies, which may occur following further development of the Hunter estuary or its catchment. The certainty of food supplies is greater when several invertebrate species are available but declines in key prey species can have drastic consequences for some shorebirds (see discussion in 8.1.1). Foraging success is often not equal among shorebirds in the same population because of gender and age effects (e.g., Stillman *et al.* 2000; Zharikov & Skilleter 2002) (see section 5.4.3), or among species, due to different foraging strategies (i.e., specialist rather than a generalist), therefore, some birds are likely to be more vulnerable to habitat modification than others.

8.1.4 Importance of saltmarsh

Coastal saltmarshes support migratory shorebirds in four ways by providing: major night roosting habitat (see discussion in 8.1.2), low and high tide feeding habitat for small species, staging habitat for passage migrants, and supplementary high tide habitat for some species during storms, spring high tides or heavy disturbance at day roosting habitat. Saltmarshes in the Hunter estuary could provide continuous feeding habitat for Common Greenshanks *Tringa nebularia*, Marsh Sandpipers *Tringa stagnatilis*, Red-necked Stints *Calidris ruficollis* and internationally significant numbers of Sharp-tailed Sandpipers *Calidris acuminata* (see Chapters 6 and 7). There were benefits to remaining in saltmarsh throughout the tidal period, as it was relatively disturbance free, food was readily available and sometimes abundant, and birds did not have to expend extra energy travelling to separate day and night roosts. Although not a substitute for the conservation of intertidal mudflats, coastal saltmarshes do have the potential to buffer migratory shorebird populations from the effects of habitat loss in Australia and other parts of the flyway.

8.1.5 Rehabilitating shorebird habitat

Managing urban wetlands can be challenging because there are often conflicting issues and management priorities. Shorebird habitat requirements can sometimes conflict with other management objectives, including the habitat requirements of some threatened species, fisheries and mosquito control. Freshwater wetlands on Ash Island (Fig. 8.1), for example, support the vulnerable Green and Golden Bell Frog *Litoria aurea* (*EPBC Act*) (Hamer *et al.* 2002), but the reintroduction of the tidal regime to restore estuarine vegetation threatens freshwater habitat. Although mangrove trees are protected in NSW (State Environmental and Planning Policy 14 - coastal wetlands (SEPP 14) and *Fisheries Management Act 1994*), the encroachment of mangrove into saltmarsh is now recognised as a major threat to shorebird habitat in south-eastern Australia (Saintilan 2003; Straw & Saintilan 2005). Further, runnelling (the construction of channels to increase tidal flushing) is often used to inhibit mosquito development by removing pools of standing water in wetlands near urban areas (Breitfuss 2003), but this technique may reduce the availability of roosting and feeding habitat for shorebirds, which prefer to roost and forage in or around the edge of shallow water. Applying the mosquito selective pesticide Bti *Bacillus thuringiensis* var. *israelensis* to urban wetlands can also impact non-target invertebrates such as chironomids (Dickman 2000; Morgan 2008), which may reduce food availability for shorebirds.

The Kooragang Wetland Rehabilitation Project (KWRP) manages urban wetlands and so faces many of these challenges. As part of its rehabilitation of wetlands on Ash Island the project supported studies of vegetation (Streever *et al.* 1996; MacDonald 2001; Nelson 2006), fish (Williams *et al.* 1995) and waterbirds (Kingsford *et al.* 1998) before and after tidal manipulation. The opportunity to monitor the success of a rehabilitation project over a ten-year time frame is rare in restoration ecology but it proved difficult to create an ideal experimental design in such a dynamic system. Not only did mangroves expand rapidly (about 1 ha yr^{-1} in area) (Howe 2008) but migratory shorebird populations were also highly variable in space and time, reducing the power to detect positive or negative changes following wetland manipulation. Greater tidal flushing initially increased habitat availability for shorebirds on Ash Island but subsequent mangrove encroachment negated these effects and within ten years there was a 17% decrease (from 31.5 to 26.1 ha) in shorebird habitat in Area E (Howe 2008). Rapid mangrove expansion coincided with increases in mean annual water level in the south arm of the Hunter River (Fig. 8.1), which were attributed to eustatic sea level rise, flood mitigation works and port development, and the effects of a 18.6 year lunar nodal cycle (Howe 2008). The extent to which mangrove

outcompeted saltmarsh was unexpected but steps have already been taken to address this imbalance by removing mangroves from shorebird habitat (see Chapter 7) and further monitoring should help support an adaptive management approach to managing mangrove and shorebird habitat on Ash Island and other parts of the estuary (see section 8.2).

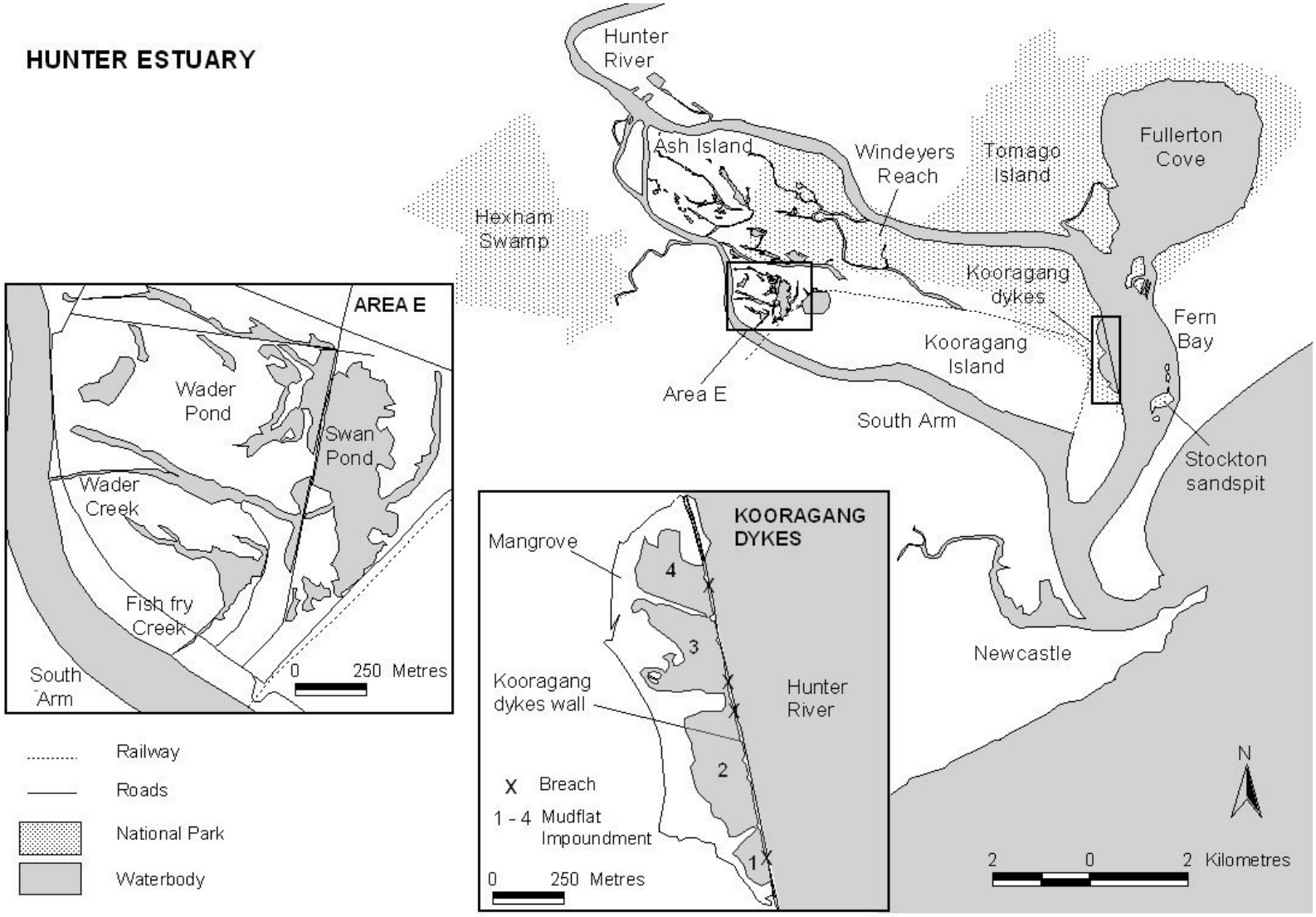


Figure 8.1 Location of major roosting (Windeyers Reach, Kooragang dykes and Stockton sandspit) and foraging (Fullerton Cove, Fern Bay, Kooragang dyke impoundments and Area E, on Ash Island) habitats for migratory shorebirds in the Hunter estuary, NSW.

8.2 Implications for management

The main threats to shorebird habitat in the Hunter estuary are further port development, human-induced disturbance, eustatic sea level rise and mangrove encroachment. The scale of habitat lost historically to port development in the Hunter estuary is staggering, yet further dredging is already underway and new development proposals threaten the viability of remaining shorebird habitat. To address this imbalance, there first needs to be greater commitment to the protection and management of shorebird habitat in this non-breeding site. Management should take an active role to maximise shorebird habitat availability by: (1) protecting high priority habitats from development, (2) actively managing disturbance at the day roosts, (3) rehabilitating existing roosts, (4) restoring saltmarsh, and (5) monitoring migratory shorebird populations and their habitats.

8.2.1 Priority habitats

High tide roosting habitat, intertidal mudflats, saltmarshes and freshwater wetlands should be recognised as priority habitats for migratory shorebirds. These habitats need to be managed using an adaptive management framework which incorporates future hydrological changes from developments and predicted sea level rise. High priority habitats in the Hunter estuary include: major day roosting habitat at the Kooragang dykes and Stockton sandspit; intertidal mudflats in Fullerton Cove, Kooragang dyke impoundments and along the lower estuary foreshore (Fern Bay); and saltmarsh-tidal pool complexes at Windeyers Reach and Area E on Kooragang Island (Fig. 8.1). All of these habitats, except Area E, are listed under the Hunter Wetlands National Park (Fig. 8.1) and Hunter Estuary Ramsar site (RIS 2002), however, Area E also fulfils the requirements of Ramsar listing (Ramsar 2009), given that it contains threatened species and ecological communities, it plays a role in the life cycle of up to 21 migratory shorebird species (see Appendix C) and it can regularly support more than 1% of the Sharp-tailed Sandpiper's flyway population. Freshwater wetlands on Ash Island also provide habitat for some species, including the Latham's Snipe *Gallinago hardwickii* (see section 7.4.1 and Appendix E2).

8.2.2 Roost management

Disturbance is a significant issue for shorebirds at day roosts (see section 3.4.3), but there are several management options that could help. Raising public awareness of the importance of Stockton sandspit and the Kooragang dykes for shorebirds should be the first step. Howe (2008) recommended promoting shorebirds to the local community to foster custodianship of these roost sites, but high tide patrols during periods of high recreational use (i.e., weekends and public holidays) could also be trialled. Although there is good signage at both day roosts detailing the importance of these sites for shorebird species, a compliance presence would increase public awareness. There would also be benefits from supporting the original proposal to construct a boardwalk and viewing hide east of the main lagoon at the sandspit (Fig. 3.2) (Svoboda 1996) where birds could be observed easily without being disturbed. If these strategies are not effective, the next step would be to fence the area to prevent public and ground predators accessing the sandspit.

Most of the people-induced disturbances at the Kooragang dykes were caused by boat traffic (Table 3.5). This was part of background noise at the Kooragang dykes and usually birds did not flush in response to passing boats, but birds were more likely to flush during spring high tides or when boats were large, or passed inside navigational markers (<40 m from the dyke wall), as this caused waves to cover the dyke wall, displacing roosting birds. Introducing buffer zones that prohibit boats travelling, drifting or mooring closer than 40 m from the Kooragang dyke wall would reduce disturbance at this day roost by 50%.

The major day and night roosts also need remedial works. At Stockton sandspit mangrove seedlings and weeds are currently removed by hand by members of the Hunter Bird Observers Club (HBOC), but day roosting habitat at the Kooragang dykes and night roosting habitat at Windeyers Reach (Fig. 8.1) are threatened by mangrove encroachment (see discussion in 8.2.3). Howe (2008) recommended further on-ground works be undertaken at Stockton sandspit including the raising of the weir and strengthening of the perimeter bund to exclude mangrove propagules and manage erosion at this site, but tidal flushing is needed to maintain saltmarsh, sandflat and open water microhabitats for roosting shorebirds (see sections 3.4.1 and 8.1.2). Howe (2008) also predicted that under current physical and chemical erosion rates the Kooragang dykes system would not provide viable roosting habitat within the next 15 - 30 years (depending

on the extent of sea level rise). The Kooragang dyke wall needs to be repaired to increase roosting habitat but major breaches in the wall (Fig. 8.1) should be preserved to ensure that tidal connectivity is maintained and the impoundments still function as foraging habitat. Dividing walls between the impoundments (Fig. 8.1) provided roosting habitat for shorebirds in former years (A. Lindsey (HBOC) *pers. comm.* 2004) but have since been invaded by mangroves. Removing mangroves from these dividing walls has great potential for increasing roosting habitat availability for shorebirds in the Hunter estuary, as these areas are already close to existing day roosting habitat and are protected from boat activity in the main channel. Fox baiting is also needed on Kooragang Island to reduce the risk of shorebird predation or disturbance in night-roosting habitat.

8.2.3 Saltmarsh restoration

Tidal reinstatement restores saltmarshes, but in southern Australia it does not always guarantee successful rehabilitation of wetlands because of imbalances in saltmarsh-mangrove dynamics (Saintilan & Williams 1999; Wilton 2002). On Ash Island the largest concentrations of shorebirds were in open areas of saltmarsh (mainly salt couch *Sporobolus virginicus* and samphire *Sarcocornia quinqueflora* < 150 mm high) that surrounded shallow tidal pools with few mangroves (see section 7.4). Most shorebird species prefer open habitats with shallow water or wet substrates not deeper than their thigh (Dann 1987; Skagen & Knopf 1994; Parsons 2002) (see Chapter 4) and not closer than 10 - 40 m from tall vegetation (Lawler 1996), allowing birds a clear view of potential predators. If restoring saltmarsh for shorebirds is the primary management objective, water levels need to be managed to maintain shallow ponds with short sparse vegetation, while preventing the establishment of mangroves.

The availability of wet substrate or shallow water in saltmarsh is determined by tidal range, topography, elevation, hydroperiod, local rainfall and evaporation (Howe 2008). Howe (2008) demonstrated that it is possible to maintain saltmarsh in lower tidal frames (tidal range <0.3 m and hydroperiod >0.3) in the Hunter estuary when infrastructure is a barrier to the upslope migration of saltmarsh communities. In Area E culverts maintained saltmarsh in lower tidal frames in Swan (wetland 16) and Wader (wetland 19) ponds (Fig. 8.1), and these areas supported the most shorebirds on Ash Island during high and low tide periods, but culvert removal favoured mangrove encroachment into wetlands connected to Fish Fry Creek (Creek 5) (Fig. 8.1) reducing habitat availability for shorebirds (see Chapter 7) (Howe *et al.* in press). Although mangroves

have since been removed from parts of Area E (see Figs. 7.1 and 7.2) to restore shorebird habitat, Howe (2008) also recommended installing a broad-crested weir or box culvert at Fish Fry Creek and closing Wader Creek (Creek 4) (Fig. 8.1) to reduce inflows and inhibit mangrove survival and propagule establishment. Although the installation of artificial structures does not provide the long-term benefits of estuary and catchment-scale approaches to managing estuarine vegetation distribution, this may be the only option for conserving saltmarsh in Area E if development of the Hunter estuary continues and global mean sea level rise predictions (0.18 – 0.59 m) for this century (IPCC 2007) are realised.

Tidal flushing has also recently been reinstated to Tomago Island (24 Sept 2008) and Hexham Swamp (19 Dec 2008) (Fig. 8.1) providing further opportunities for gains in saltmarsh in the Hunter estuary. Both sites lost saltmarsh (87 and 99%, respectively) (Williams *et al.* 2000; HCRCMA 2009) following the construction of flood gates, levee banks and ring drains in the 1970s for flood mitigation (see historical review in section 2.2). Tomago Island has great potential to provide new roosting and supplementary feeding habitat for shorebirds as this site historically provided night roosting habitat (Clarke & van Gessel 1983) and it is close to feeding habitat in Fullerton Cove. Rehabilitated sites on Tomago Island, Hexham Swamp and Kooragang Island will require ongoing monitoring and management intervention to prevent mangrove dominating newly flooded habitats and to minimise the risk of leaching from acid sulphate soils identified at these sites (DPI 2008).

Although I have emphasised the importance of saltmarsh as feeding and roosting habitat for shorebirds and the threat of mangrove encroachment into saltmarsh, it is not necessary to remove all mangroves to maximise shorebird habitat as large stands of mangroves (with trees >5 m in height above water) do provide roosting habitat for some shorebirds (e.g., small flocks of Grey-tailed Tattlers *Heteroscelus brevipes* and Whimbrels) (Lawler 1996) and nursery habitat for fish and crustaceans (Robertson & Duke 1987; Mazumder *et al.* 2006). The most important principle for managing shorebird habitat is to remove tall vegetation (>5 m) which is within 40 m of roosting shorebirds (see Lawler 1996). Under the NSW *Fisheries Management Act 1994* new provisions allow the removal of mangroves where they are degrading shorebird habitat (e.g., Stockton sandspit and Ash Island in the Hunter estuary) (see Chapter 7). Further remedial works are needed to remove mangroves fringing ponds at Windeyers Reach to ensure that night roosting

habitat remains viable at this site. These works should be undertaken in winter months (May-early August) to ensure the least disturbance to migratory shorebirds.

8.2.4 Monitoring shorebirds and their habitats

Long-term funding is required to support monitoring of shorebird populations and their habitats in the Hunter estuary. Shorebird surveys are currently done on a voluntary basis by a small team of dedicated enthusiasts from the HBOC, which have provided valuable count data, but this survey effort needs to be supported for it to persist in the long-term. Regular counts facilitate greater understanding of shorebird population dynamics and can be used to measure the effectiveness of management actions (e.g., recent tidal reinstatement on Tomago Island and Hexham Swamp). A long-term monitoring program measuring the extent of mangrove, intertidal mudflat and saltmarsh is also needed to direct rehabilitation efforts. As mangrove can respond rapidly to increases in water level (see section 8.2.3), analysis of aerial photography may be required on a two to three year basis to identify pressures on saltmarsh.

8.2.5 Summary of recommendations

My main recommendations for managing estuarine habitats for migratory shorebirds in the Hunter estuary are as follows (see also Spencer & Howe 2008):

1. Protect key roosting and foraging habitats for shorebirds in the Hunter estuary from development (see section 8.2.1).
2. Formally recognise Area E, on Ash Island, as a significant wetland for migratory shorebirds under the Hunter Wetlands National Park and Ramsar site (see supporting evidence in Appendix C).
3. Promote the importance of Stockton sandspit and the Kooragang dykes as roosting habitat for shorebirds to the local community.
4. Install a new hide and boardwalk east of the main lagoon at the sandspit (Svoboda 1996) so that shorebirds can be observed without being disturbed and to attract visitors to the site.
5. Conduct regular high tide patrols of the Stockton sandspit and Kooragang dykes during weekends and public holidays to facilitate public awareness of the importance of these sites for roosting shorebirds.

6. Establish buffer distances from the Kooragang dykes that prohibit boats travelling, drifting or mooring closer than 40 m from the dyke wall to reduce disturbance of roosting shorebirds.
7. Conduct remedial works to manage erosion at Stockton sandspit (see recommendations in Howe (2008)).
8. Repair eroding roosting habitat on the Kooragang dykes without modifying major breaches in the dyke wall (see Fig. 8.1) to ensure tidal connectivity is maintained to the mudflat impoundments, which provide foraging habitat for shorebirds.
9. Remove mangroves from walls dividing the Kooragang dyke impoundments (see Fig. 8.1) to create new roosting habitat.
10. Remove mangroves from saltmarsh/open water complexes at Windeyers Reach to maintain night roosting habitat. This work should be conducted in winter months (May – early August), when migratory shorebird numbers are low, to minimise disturbance.
11. Conduct fox baiting on Kooragang Island.
12. Restrict inflows through Wader and Fish Fry creeks (creeks 4 and 5) on Ash Island to promote the establishment of saltmarsh but inhibit mangrove expansion in Area E (see recommendations in Howe (2008)).
13. Support long-term monitoring of shorebird populations and further studies of shorebirds and their habitats in the Hunter estuary (see section 8.3).

8.3 Scope for further research

Further research of migratory shorebird ecology is needed in the Hunter estuary and other parts of their non-breeding range to advance our understanding of shorebird habitat use and threats to key habitats. Potential studies that would follow on from this research are:

1. Locate and map all nocturnal roost sites in the Hunter estuary. This is a priority for managing roost habitat for shorebirds in the estuary. This could be achieved through a combination of radio-tracking studies and explorative night surveys of saltmarsh.
2. Undertake radio-telemetry studies to identify important foraging and night roosting habitat for small shorebird species which have suffered population declines, including Curlew Sandpipers and Pacific Golden Plovers, or for species whose population status and roost sites are unknown, such as the Double-banded Plover *Charadrius bicinctus*.
3. Identify key foraging habitats and nocturnal roost sites for the threatened Black-tailed Godwit (*NSW TSC Act 1995; IUCN Red List*). The Hunter estuary is a stronghold for Black-tailed Godwits in NSW (Smith 1991) but total counts of this species have declined by 44% in the Hunter estuary since the 1980s (see Chapter 2).
4. Investigate benthic invertebrate assemblages in intertidal mudflats in the Hunter estuary to increase our understanding of the natural temporal and spatial variability associated with invertebrate populations and their vulnerability to habitat modification from changes to the estuary and upper catchment. The minimum sampling needed to document the biotic and abiotic variability of Fullerton Cove, for example, would be bi-monthly for two years (Hutchings 1983).
5. Investigate relationships between prey and common shorebird species to identify which invertebrate species are most important. Stable isotope analysis (Mizutani *et al.* 1990; Kwak & Zedler 1997) could be used to investigate links between shorebirds and their food supplies.
6. Investigate the contribution of saltwater mosquitoes to shorebird diets. This research has important implications for mosquito control programs in urban wetlands.
7. Investigate how water levels in saltmarsh can be manipulated to maximise chironomid abundance and food availability for shorebirds.
8. Conduct further low and high tide surveys of Ash Island to determine how successful mangrove removal has been in the restoration of saltmarsh habitat for shorebirds.

REFERENCES

Adam, P. (1981). Australian Saltmarshes. *Wetlands (Australia)* **1**(1), 8-10.

Adam, P. (2002). Saltmarshes in a time of change. *Environmental Conservation* **29**(1), 39-61.

Altmann, J. (1974). Observational study of behavior: sampling methods. *Behaviour* **49**, 227-267.

Amat, J. A., and Masero, J. A. (2007). The functions of belly-soaking in Kentish Plovers *Charadrius alexandrinus*. *Ibis* **149**, 91-97.

Amlaner, C. J., and McFarland, D. J. (1981). Sleep in the Herring Gull (*Larus argentatus*). *Animal Behaviour* **29**, 551-556.

Atkinson, P. W. (2003). Can we recreate or restore intertidal habitats for shorebirds? *Wader Study Group Bulletin* **100**, 67-72.

Atkinson, P. W., Crooks, S., Drewitt, A., Grant, A., Rehfisch, M. M., Sharpe, J., and Tyas, C. J. (2004). Managed realignment in the UK - the first five years of colonisation by birds. *Ibis* **146**, 101-110.

Atkinson, P. W., Baker, A. J., Bevan, R. M., Clarke, N. A., Cole, K. B., Gonzalez, P. M., Newton, J., Niles, L. J., and Robinson, R. A. (2005). Unravelling the migratory strategies of a long-distance migrant using stable isotopes: Red Knot *Calidris canutus* movements in the Americas. *Ibis* **147**(4), 738-749.

Austin, G. E., and Rehfisch, M. M. (2003). The likely impact of sea level rise on waders (Charadrii) wintering on estuaries. *Journal for Nature Conservation* **11**(1), 43-58.

Baker, A. J., Gonzalez, P. M., Piersma, T., Niles, L. J., do Nascimento, I. D. S., Atkinson, P. W., Clark, N. A., Minton, C. D. T., Peck, M. K., and Aarts, G. (2004). Rapid population decline in red knots: fitness consequences of decreased refuelling rates and late arrival in Delaware Bay. *Proceedings of the Royal Society of London Series B-Biological Sciences* **271**, 875-882.

Bamford, M. (1992). Evidence of human predation on waders in EAASRN. *The Stilt* **20**, 38-40.

Bamford, M., Watkins, D., Bancroft, W., Tischler, G., and Wahl, J. (2008). Migratory Shorebirds of the East Asian-Australasian Flyway: Population Estimates and Internationally Important Sites. (Wetlands International-Oceania: Canberra).

Barbosa, A. (1997). Foraging strategy and predator avoidance behaviour: an intraspecific approach. *Acta Oecologica-International Journal of Ecology* **18**(6), 615-620.

Barrett, G., Silcocks, A., Barry, S., Cunningham, R., and Poulter, R. (2003). 'The New Atlas of Australian Birds'. (Birds Australia: Melbourne).

Barter, M., and Minton, C. (1987). Biometrics, moult and migration of Double-banded Plovers *Charadrius bicinctus bicinctus* spending the non-breeding season in Victoria. *The Stilt* **10**, 9-14.

Barter, M. (1988). Biometrics and moult of Lesser Golden Plovers *Pluvialis dominica fulva* in Victoria. *The Stilt* **13**, 15-19.

Barter, M., Jessop, A., and Minton, C. (1988). Red Knot *Calidris canutus rogersi* in Australia. Part 2: Biometrics and moult in Victoria and North-western Australia. *The Stilt* **13**, 20-27.

Barter, M. (1989a). Bar-tailed Godwit *Limosa lapponica* in Australia. Part 1: Races, breeding areas and migration routes. *The Stilt* **14**, 43-48.

Barter, M. (1989b). Bar-tailed Godwit *Limosa lapponica* in Australia. Part 2: Weight, moult and breeding success. *The Stilt* **14**, 49-53.

Barter, M. (1992). Changing wader numbers in Swan Bay, Victoria - a cause for concern? *The Stilt* **21**, 8-12.

Barter, M., Fawen, Q., Sixian, T., Xiao, Y., and Tonkinson, D. (1997). Hunting of migratory waders on Chongming Dao: a declining occupation? *The Stilt* **31**, 18-22.

Barter, M. (2002). Shorebirds of the Yellow Sea: Importance, threats and conservation status. Wetlands International Global Series 9, International Wader Studies 12. (Wetlands International: Canberra).

Bates, B. C., Kundzewicz, S., Wu, S., and Palutikof, J. P. (2008). Climate Change and Water. Technical Paper of the Intergovernmental Panel on Climate Change. (IPCC Secretariat: Geneva).

Battley, P. F. (1997). The Northward Migration of Arctic Waders in New Zealand: Departure Behaviour, Timing and Possible Migration Routes of Red Knots and Bar-tailed Godwits from Farewell Spit, North-west Nelson. *Emu* **97**, 108-120.

Battley, P. F. (2002). Behavioural ecophysiology of migrating Great Knots. Ph.D. thesis, Australian School of Environmental Studies, Griffith University, Brisbane.

Battley, P. F., Rogers, D. I., Piersma, T., and Koolhaas, A. (2003). Behavioural evidence for heat-load problems in Great Knots in tropical Australia fuelling for long-distance flight. *Emu* **103**(2), 97-103.

Battley, P. F., Piersma, T., Rogers, D. I., Dekinga, A., Spaans, B., and van Gils, J. A. (2004). Do body condition and plumage during fuelling predict northward departure dates of Great Knot *Calidris tenuirostris* from north-west Australia. *Ibis* **146**, 46-60.

Battley, P. F., and Piersma, T. (2005). Body composition and flight ranges of Bar-tailed Godwits (*Limosa lapponica baueri*) from New Zealand. *The Auk* **122**(3), 922-937.

Bayliss, P., and Yeomans, K. M. (1990). Use of Low-level Aerial Photography to Correct Bias in Aerial Survey Estimates of Magpie Goose and Whistling Duck Density in the Northern Territory. *Australian Wildlife Research* **17**, 1-10.

Beeton, R., Buckley, K., Jones, G., Morgan, D., Reichelt, R., and Trewin, D. (2006). Australian State of the Environment 2006: Independent report to the Australian Government Minister for the Environment and Heritage. (Department of the Environment and Heritage: Canberra).

Bildstein, K., Christy, R., and DeCoursey, P. (1982). Energy flow through a South Carolina saltmarsh shorebird community. *Wader Study Group Bulletin* **34**, 35-37.

Birding-Taiwan (2008). Available at: http://www.birdingintaiwan.com/ (accessed 18 July 2008).

Birds-Korea (2008). Birds Korea National Shorebird Survey 2008. Available at: http://www.birdskorea.org (accessed 18 July 2008).

Blakey, R., Zharikov, Y., and Skilleter, G. A. (2006). Lack of an osmotic constraint on intake rate of the Eastern Curlew *Numenius madagascariensis*. *Journal of Avian Biology* **37**(4), 299-305.

Blumstein, D. T., Anthony, L. L., Harcourt, R., and Ross, G. (2003). Testing a key assumption of wildlife buffer zones: is flight initiation distance a species-specific trait? *Biological Conservation* **110**, 97-100.

BOM (2004). National Tidal Facility, Commonwealth of Australia Bureau of Meteorology. Available at: http://www.bom.gov.au/oceanography/tides/ (accessed September 2004).

BOM (2005). National Tidal Facility, Commonwealth of Australia Bureau of Meteorology. Available at: http://www.bom.gov.au/oceanography/tides/ (accessed September 2005).

BOM (2007). Newcastle University, New South Wales, Daily Weather Observations. Available at: http://www.bom.gov.au/climate/dwo/IDCJDW2098.latest.shtml (accessed April 2007).

Bowen, R., Stephens, N., and Donnelly, P. (1995). SEPP 14 - Wetland Protection and the Role of Mitigation. *Wetlands (Australia)* **14**, 6-12.

Braithwaite, L. W., Maher, M., Briggs, S. V., and Parker, B. S. (1986). An Aerial Survey of Three Game Species of Waterfowl (Family Anatidae) Populations in Eastern Australia. *Australian Wildlife Research* **13**, 213-223.

Bray, J. R., and Curtis, J. T. (1957). An ordination of the upland forest communities of southern Wisconsin. *Ecological Monographs* **27**, 325-349.

Brayton, A. F., and Schneider, D. C. (2000). Shorebird abundance and invertebrate density during the boreal winter and spring at Peninsula Valdez, Argentina. *Waterbirds* **23**(2), 277-282.

Breitfuss, M. (2003). The effects of physical habitat modification for mosquito control, runnelling, on selected non-target saltmarsh resources. Ph.D. Thesis, Australian School of Environmental Studies, Griffith University, Brisbane.

Brinson, M. M., and Rheinhardt, R. (1996). The Role of Reference Wetlands in Functional Assessment and Mitigation. *Ecological Applications* **6**(1), 69-76.

Brown, S., and Lugo, A. E. (1994). Rehabilitation of Tropical Lands: A Key to Sustaining Development. *Restoration Ecology* **2**(2), 97-11.

Bucher, D., and Saenger, P. (1991). An Inventory of Australian Estuaries and Enclosed Marine Waters: An Overview of Results. *Australian Geographical Studies* **29**, 370-381.

Buckney, R. T. (1987). Three Decades of Habitat Change: Kooragang Island, New South Wales. In 'Nature Conservation: The role of remnants of native vegetation'. (Ed. D. A. Saunders, G. W. Arnold, A. A. Burbridge, and A. J. M. Hopkins). pp. 227-232. (Surrey Beatty and Sons: Chipping Norton, NSW).

Bunn, S., and Arthington, A. (2002). Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. *Environmental Management* **30**(4), 492-507.

Burchett, M. D., Pulkownik, A., Grant, C., and Macfarlane, G. (1998). Rehabilitation of Saline Wetlands, Olympics 2000 Site, Sydney (Australia) - I: Management Strategies Based on Ecological Needs Assessment. *Marine Pollution Bulletin* **37**(8-12), 515-525.

Burger, J., Howe, M. A., Hahn, D. C., and Chase, J. (1977). Effects of tide cycles on habitat selection and habitat partitioning by migrating shorebirds. *The Auk* **94**, 743-758.

Burger, J. (1981). The effect of human activity on birds at a coastal bay. *Biological Conservation* **21**, 231-241.

Burger, J., and Gochfeld, M. (1991). Human activity influence and diurnal and nocturnal foraging of sanderlings (*Calidris alba*). *The Condor* **93**, 259-265.

Burger, J. (1997). Importance of beach, mudflat and marsh habitats to migrant shorebirds on Delaware Bay. *Biological Conservation* **79**, 283-292.

Burton, A. C. G., and Morris, A. K. (1993). New South Wales Annual Bird Report - 1990. *Australian Birds* **26**(4), 89-120.

Burton, N. H. K., Evans, P. R., and Robinson, M. A. (1996). Effects on Shorebird Numbers of Disturbance, the Loss of a Roost Site and its Replacement by an Artificial Island at Hartlepool, Cleveland. *Biological Conservation* **77**, 193-201.

Burton, N. H. K., Armitage, M. J. S., Musgrove, A. J., and Rehfisch, M. M. (2002a). Impacts of man-made landscape features on numbers of estuarine waterbirds at low tide. *Environmental Management* **30**(6), 857-864.

Burton, N. H. K., Rehfisch, M. M., and Clark, N. A. (2002b). Impacts of disturbance from construction work on the densities and feeding behaviour of waterbirds using the intertidal mudflats of Cardiff Bay, UK. *Environmental Management* **30**(6), 865-871.

Burton, N. H. K., Rehfisch, M. M., Clark, N. A., and Dodd, S. G. (2006). Impacts of sudden winter habitat loss on the body condition and survival of redshank *Tringa totanus*. *Journal of Applied Ecology* **43**, 464-473.

Caldow, W. G., Goss-Custard, J. D., Stillman, R. A., Dit Durell, S. E. A. L. V., Swinfen, R., and Bregnballe, T. (1999). Individual variation in the competitive ability of interference-prone foragers: the relative importance of foraging efficiency and susceptibility to interference. *Journal of Animal Ecology* **68**, 869-878.

Callaway, J. C., and Josselyn, M. N. (1992). The introduction and spread of smooth cordgrass (*Spartina alterniflora*) in south San Francisco Bay. *Estuaries* **15**, 218-226.

Chambers, L. E., Hughes, L., and Weston, M. A. (2005). Climate change and its impact on Australia's avifauna. *Emu* **105**(1), 1-20.

Chatto, R. (2005). A Tour of Selected Significant Coastal Shorebird Sites in the Top End of the Northern Territory of Australia. In 'Status and Conservation of Shorebirds in the East Asian-Australasian Flyway; Proceedings of the Australasian Shorebird Conference 13-15 December 2003, Canberra, Australia.' (Ed. P. Straw). pp. 44-51. (Australasian Wader Studies Group and Wetlands International - Oceania.).

Clarke, C. J., and van Gessel, F. W. C. (1983). Habitat Evaluation - Birds. In 'An investigation of the natural areas of Kooragang Island, Hunter River.'. (Ed. J. Moss). pp. 117-144. (Department of Environment and Planning: Sydney).

Clarke, K. R., and Ainsworth, M. (1993). A method of linking multivariate community structure to environmental variables. *Marine Ecology Progress Series* **92**, 205-219.

Clarke, K. R., and Warwick, R. M. (2001). 'Changes in marine communities: An approach to statistical analysis and interpretation'. 2nd edition. (PRIMER-E: Plymouth, United Kingdom).

Clarke, P. J., and Myerscough, P. J. (1993). The intertidal distribution of the grey mangrove (*Avicennia marina*) in southeastern Australia: The effects of physical conditions, interspecific competition, and predation on propagule establishment and survival. *Australian Journal of Ecology* **18**, 307-315.

Clemens, R. S., Haslem, A., Oldland, J. M., Weston, M. A., Spencer, J., Milton, D., Rogers, D., Rogers, K., Gosbell, K., Ferris, J., and Bamford, M. (2008). A Population Monitoring Program for Shorebirds in Australia. Draft report prepared for the Australian Government's Department of Environment, Water, Heritage and Arts. (Birds Australia: Melbourne).

Close, D., and Newman, O. M. G. (1984). The decline of the Eastern Curlew in South-eastern Australia. *Emu* **84**, 38-40.

Coffey, E. J. (1973). Report and Findings of the Commissioner Inquiry into Pollution from Kooragang Island. (State Pollution Control Commission: Sydney).

Coleman, R. A., Salmon, N. A., and Hawkins, S. J. (2003). Sub-dispersive human disturbance of foraging oystercatchers *Haematopus ostralegus*. *Ardea* **91**(2), 263-268.

Colwell, M. A., and Mathis, R. L. (2001). Seasonal variation in territory occupancy of non-breeding Long-billed Curlews in intertidal habitats. *Waterbirds* **24**(2), 208-216.

Colwell, M. A., Danufsky, T., Fox-Fernandez, N. W., Roth, J. E., and Conklin, J. R. (2003). Variation in shorebird use of diurnal, high-tide roosts: How consistently are roosts used? *Waterbirds* **26**(4), 484-493.

Conners, P. G., Myers, J. P., Conners, C. S. W., and Pitelka, F. A. (1981). Interhabitat movements by sanderlings in relation to foraging profitability and the tidal cycle. *The Auk* **98**, 49-64.

Connolly, R. M. (1994). A Comparison of Fish Assemblages from Seagrass and Unvegetated Areas of a Southern Australian Estuary. *Australian Journal of Marine and Freshwater Research* **45**, 1033-1044.

Coopers, R. M. (1989). New South Wales Bird Report - 1985. *Australian Birds* **22**(1-2), 1-40.

Coopers, R. M. (1990). New South Wales Annual Bird Report 1986. *Australian Birds* **23**(4), 68-101.
Coopers, R. M. (1991). New South Wales Bird Report - 1987. *Australian Birds* **24**(3), 49-72.
Coopers, R. M. (1992). New South Wales Bird Report - 1988. *Australian Birds* **25**(4), 85-99.
Craven, P., Worrell, R., Hall, P., Corrigan, V., and Jarman, M. (2006). Sharing the shoreline: shorebird population monitoring and habitat management at the Shoalhaven River Estuary, South Coast, New South Wales, Australia. *The Stilt* **49**, 56.
Crawford, L., and Herbert, C. (2007). Pacific Golden Plovers in the Hunter Estuary. Hunter Bird Observers Club. (Avifauna Research and Services: Sydney).
Creed, K. E., and Bailey, M. (1998). Decline in migratory waders at Pelican Point, Swan River, Western Australia. *The Stilt* **33**, 10-14.
Cresswell, W. (1994). Age-Dependent Choice of Redshank (*Tringa totanus*) Feeding Location: Profitability or Risk? *The Journal of Animal Ecology* **63**(3), 589-600.
Dahl, T. E. (2000). Status and Trends of Wetlands in the Conterminous United States 1986 to 1997. (US Department of the Interior, Fish and Wildlife Service: Washington, D.C.).
Dames, and Moore. (1978). An assessment of the effect on the environment of the proposed Stage II Land Fill Scheme at Kooragang Island, Newcastle, New South Wales. (New South Wales Department of Public Works: Sydney).
Dann, P. (1981a). The evidence for feeding ground fidelity of waders in southern Victoria. *VWSG Bulletin* **3**, 13-14.
Dann, P. (1981b). Resource allocation among three congeneric species of sandpiper. *The Stilt* **1**, 3.
Dann, P. (1983). Feeding behaviour of four species of Calidridine sandpiper at Lake Reeve in East Gippsland. *The Victorian Wader Study Group Bulletin* **7**, 2-6.
Dann, P. (1987). The feeding behaviour and ecology of shorebirds. In 'Shorebirds in Australia'. (Ed. B. Lane). pp. 10-20. (Nelson: Melbourne).
Dann, P. (1999a). Foraging behaviour and diets of red-necked stints and curlew sandpipers in south-eastern Australia. *Wildlife Research* **27**, 61-68.
Dann, P. (1999b). Short Communication: Feeding Periods and Supratidal Feeding of Red-necked Stints and Curlew Sandpipers in Western Port, Victoria. *Emu* **99**, 218-222.
Dann, P. (1999c). Feeding ecology of Eastern Curlew in Western Port Bay, Victoria. *The Stilt* **15**, 60-61.
Dann, P. (2000a). Feeding Behaviour and Diet of Double-banded Plovers *Charadrius bicinctus* in Western Port, Victoria. *Emu* **91**(3), 179-184.
Dann, P. M. (2000b). The ecology of birds frequenting intertidal habitats in Western Port, Victoria. Ph.D. Thesis, Department of Zoology, University of Melbourne, Australia.
Darnell, T. M., and Smith, E. H. (2004). Avian use of natural and created salt marsh in Texas, USA. *Waterbirds* **27**(3), 355-361.
de Boer, W. F., and Longamane, F. A. (1996). The exploitation of intertidal food resources in Inhaca Bay, Mozambique, by shorebirds and humans. *Biological Conservation* **78**, 295-303.
DECC (2004). Coastal saltmarsh in the NSW North Coast, Sydney Basin and South East Corner bioregions - endangered ecological community listing. NSW Scientific Committee - final determination. Available at: http://www.environment.nsw.gov.au/determinations/CoastalSaltmarshEndSpListing.htm (accessed 24 July 2008).
DECC (2007). New South Wales Department of Environment and Climate Change: The Hunter Estuary National Park. Available at:

http://www.environment.nsw.gov.au/NationalParks/parkHome.aspx?id=N1104 (accessed 18 July 2008).

DEH (2005). Background Paper to the Wildlife Conservation Plan for Migratory Shorebirds. (Department of Environment and Heritage, Canberra).

DEH (2006). Wildlife Conservation Plan for Migratory Shorebirds. (Department of Environment and Heritage: Canberra).

Dekker, D. (1998). Over-ocean flocking by Dunlins, *Calidris alpina*, and the effect of raptor predation at Boundary Bay, British Columbia. *Canadian Field-Naturalist* **112**(4), 694-697.

Dekker, D., and Ydenberg, R. (2004). Raptor predation on wintering Dunlins in relation to the tidal cycle. *Condor* **106**(2), 415-419.

Dias, M. P., Granadeiro, J. P., Lecoq, M., Santos, C. D., and Palmeirim, J. M. (2006). Distance to high-tide roosts constrains the use of foraging areas by dunlins: Implications for the management of estuarine wetlands. *Biological Conservation* **131**, 446-452.

Dickman, M. (2000). Impacts of a mosquito selective pesticide, Bti, on the macroinvertebrates of a subtropical stream in Hong Kong. *Chemosphere* **41**, 209-217.

Dittmann, S. (2002). Benthic fauna in tropical tidal flats of Hinchinbrook Channel, NE Australia: diversity, abundance and their spatial and temporal variation. *Wetlands Ecology and Management* **10**, 323-333.

Dominguez, J. (2003). Sleeping and vigilance in Black-tailed Godwit. *Journal of Ethology* **21**, 57-60.

DPI (2008). Acid Sulfate Soils Priority Investigations for the Lower Hunter River Estuary. Report to the Department of Environment, Water, Heritage and the Arts. (NSW Department of Primary Industries: Port Stephens).

Driscoll, P. (1995). Sightings of waders leg-flagged in Moreton Bay, South east Queensland. *The Stilt* **26**, 40-41.

Driscoll, P. V., and Ueta, M. (2002). The migration route and behaviour of Eastern Curlews *Numenius madagascariensis*. *Ibis* **144**, E119-E130.

Dugan, P. (1993). 'Wetlands in Danger: A World Conservation Atlas'. (Oxford University Press: New York).

Durell, S. E. A. (2000). Individual feeding specialisation in shorebirds: population consequences and conservation implications. *Biological Reviews* **75**(4), 503-518.

Durell, S. E. A., Stillman, R. A., Triplet, P., Aulert, C., Biot, D. O. D., Bouchet, A., Duhamel, S., Mayot, S., and Goss-Custard, J. D. (2005). Modelling the efficacy of proposed mitigation areas for shorebirds: a case study on the Seine estuary, France. *Biological Conservation* **123**(1), 67-77.

Ekert, P. A. (2003). Shorebird Monitoring in Kooragang Nature Reserve Stage 1. Final Report for National Parks and Wildlife Service Northern Directorate. (Ekerlogic Consulting Services: Wallsend, NSW).

Engelmoer, M., Piersma, T., Altenburg, W., and Mes, R. (1984). The Banc d'Arguin (Mauritania). In 'Coastal Waders and Wildfowl in Winter'. (Ed. P. R. Evans, J. D. Goss-Custard, and W. G. Hale). pp. 293-310. (Cambridge University Press: Cambridge, UK.).

Erwin, R. M. (1996). Dependence of Waterbirds and Shorebirds on Shallow-Water Habitats in the Mid-Atlantic Coastal Region: An Ecological Profile and Management Recommendations. *Estuaries* **19**(2), 213-219.

Evans, P. R., Herdson, D. M., Knights, P. J., and Pienkowski, M. W. (1979). Short-Term Effects of Reclamation of Part of Seal Sands, Teesmouth, on Wintering Waders and Shelduck. I.

Shorebird Diets, Invertebrate Densities, and the Impact of Predation on the Invertebrates. *Oecologia* **41**, 183-206.

Evans, P. R., Goss-Custard, J. D., and Hale, W. G. (1984). 'Coastal Waders and Wildfowl in Winter'. (Cambridge University Press, Cambridge, UK.).

Evans, P. R. (1988). Predation of intertidal fauna by shorebirds in relation to time of day, tide and year. In 'Behavioural Adaptation to Intertidal Life'. (Ed. G. Chelazzi, and M. Vannini). pp. 65-78. (Plenum Press: New York and London).

Evans, P. R., Ward, R. M., Bone, M., and Leakey, M. (1998). Creation of temperate-climate intertidal mudflats: Factors affecting colonization and use by benthic invertebrates and their bird predators. *Marine Pollution Bulletin* **37**(8-12), 535-545.

Farmer, A., Rye, R., Landis, G., Bern, C., Kester, C., and Ridley, I. (2003). Tracing the pathways of Neotropical Migratory Shorebirds using stable isotopes: a pilot study. *Isotopes in Environmental and Health Studies* **39**(3), 169-177.

Field, C. D. (1998). Rehabilitation of Mangrove Ecosystems: An Overview. *Marine Pollution Bulletin* **37**(8-12), 383-392.

Finn, P. G., Catterall, C. P., and Driscoll, P. V. (2001). The low-tide distribution of Eastern Curlew on feeding grounds in Moreton Bay, Queensland. *The Stilt* **38**, 9-17.

Finn, P. G. (2007). Feeding ecology and habitat selection. In 'Shorebirds of Australia'. (Ed. A. Geering, L. Agnew, and S. Harding). pp. 51-59. (CSIRO Publishing: Collingwood, Victoria).

Finn, P. G., Catterall, C. P., and Driscoll, P. (2007). Determinants of preferred intertidal feeding habitats for Eastern Curlew: A study at two spatial scales. *Austral Ecology* **32**, 131-144.

Fitzpatrick, S., and Bouchez, B. (1998). Effects of recreational disturbance on the foraging behaviour of waders on a rocky beach. *Bird Study* **45**, 157-171.

Foate, M. A. (2005). Summer distribution and movement of Bar-tailed Godwit (*Limosa lapponica*) in the Hunter Estuary, and their behavioural responses to disturbances at the diurnal roost sites. Hons. Thesis, Biological Sciences, University of Newcastle, Australia.

Fowler, J., Cohen, L., and Jarvis, P. (1998). 'Practical Statistics for Field Biology'. (John Wiley & Sons Ltd: Chichester, UK).

Furness, R. W. (1973). Roost selection by waders. *Scottish Birds* **7**, 281-287.

Galbraith, H., Jones, R., Park, R., Clough, J., Herrod-Julius, S., Harrington, B., and Page, G. (2002). Global climate change and sea level rise: Potential losses of intertidal habitat for shorebirds. *Waterbirds* **25**(2), 173-183.

Garbutt, R. A., Reading, C. J., Wolters, M., Gray, A. J., and Rothery, P. (2006). Monitoring the development of intertidal habitats on former agricultural land after the managed realignment of coastal defences at Tollesbury, Essex, UK. *Marine Pollution Bulletin* **53**(1), 155-164.

Garnett, S. (1986). Seasonal changes in the wader population in the south-east of the Gulf of Carpentaria. *The Stilt* **8**, 9-13.

Garnett, S. T. (1987). Aerial Surveys of Waders (Aves: Charadriiformes) Along the Coast of North-eastern Australia. *Australian Wildlife Research* **14**, 521-528.

Geering, D., and Winning, G. (1993). Waders of the Hunter River Estuary: threats to habitat and opportunities for habitat restoration. Report to Kooragang Wetland Rehabilitation Project. (Shortlands Wetland Centre: Newcastle).

Geering, D., and Winning, G. (1994). Recommended Rehabilitation Strategy for Migratory Shorebird Habitat at Stockton Sandspit. Draft report to the Kooragang Wetland Rehabilitation Project. (Shortlands Wetland Centre: Newcastle).

Geering, D. J. (1995). Ecology of migratory shorebirds in the Hunter River estuary. (Shortlands Wetland Centre: Newcastle).

GHD (2003). Proposed extension of shipping channels Port of Newcastle. Environmental Impact Statement. GHD Pty Ltd: Sydney.

Gill, B. (2006). Good News, Band News, and Great News: Satellite tagged Bar-tailed Godwits. Tattler: Newsletter for the Asia Pacific Flyways. January 2006. (Australasian Wader Study Group: Sydney).

Gill, R. E. J., Tibbitts, T. L., Douglas, D. C., Handel, C. M., Mulcahy, D. M., Gottschalck, J. C., Warnock, N., McCaffery, B. J., Battley, P. F., and Piersma, T. (2008). Extreme endurance flights by landbird crossing the Pacific Ocean: ecological corridor rather than barrier? *Proceeding of Royal Society London (B) (Online)* **276**(1656), 447-457. Available at: http://rspb.royalsocietypublishing.org/content/276/1656/447.full (accessed 25 January 2009).

Gillanders, B., and Kingsford, M. (2002). Impact of changes in flow of freshwater on estuarine and open coastal habitats and the associated organisms. In 'Oceanography and Marine Biology: an Annual Review'. (Ed. H. Barnes). pp. 233-309. (Allen and Unwin: London, UK).

Gosbell, K., and Grear, B. (2005). The Importance of Monitoring Shorebird Utilisation of the Coorong and Surrounding Wetlands in South Australia. In 'Status and Conservation of Shorebirds in the East Asian-Australasian Flyway; Proceedings of the Australasian Shorebird Conference 13-15 December 2003, Canberra, Australia.' (Ed. P. Straw). pp. 52-62. (Australasian Wader Studies Group and Wetlands International - Oceania).

Gosbell, K., and Clemens, R. (2006). Population monitoring in Australia: some insights after 25 years and future directions. *Stilt* **50**, 162-175.

Gosper, D. G. (1974). Recent records of the Black-tailed Godwit on the Hunter Estuary including a description of the breeding plumage. *NSW Birds* **8**, 62-66.

Gosper, D. G. (1981). Survey of Birds on Floodplain-estuarine Wetlands on the Hunter and Richmond Rivers in Northern NSW. *Corella* **5**(1), 1-18.

Goss-Custard, J. (1970). The responses of redshank (*Tringa totanus* (*L.*)) to spatial variations in the density of their prey. *Journal of Animal Ecology* **39**(1), 91-113.

Goss-Custard, J., and Moser, M. (1988). Rates of change in the numbers of Dunlin, *Calidris alpina*, wintering in British estuaries in relation to the spread of *Spartina anglica*. *Journal of Applied Ecology* **25**, 95-109.

Goss-Custard, J. D. (1977a). The energetics of prey selection by redshank, *Tringa totanus* (L.), in relation to prey density. *Journal of Animal Ecology* **46**(1), 1-19.

Goss-Custard, J. D. (1977b). Responses of redshank, *Tringa totanus*, to the absolute and relative densities of two prey species. *Journal of Animal Ecology* **46**(3), 867-874.

Goss-Custard, J. D., Jones, R. E., and Newbery, P. E. (1977). The Ecology of the Wash: I Distribution and diet of wading birds (Charadrii). *Journal of Applied Ecology* **14**, 681-700.

Goss-Custard, J. D., and Yates, M. G. (1992). Towards predicting the effect of salt-marsh reclamation on feeding bird numbers on the Wash. *Journal of Applied Ecology* **29**, 330-340.

Goss-Custard, J. D., Triplet, P., and West, A. D. (2006). Critical thresholds of disturbance by people and raptors in foraging wading birds. *Biological Conservation* **127**(1), 88-97.

Grayson, J. E., Chapman, M. G., and Underwood, A. J. (1999). The assessment of restoration of habitat in urban wetlands. *Landscape and Urban Planning* **43**, 227-236.

Green, R. H. (1993). Application of repeated measures design in environmental impact and monitoring studies. *Australian Journal of Ecology* **18**, 81-98.
Haig, S. M., Mehlman, D. W., and Oring, L. W. (1998). Avian Movements and Wetland Connectivity in Landscape Conservation. *Conservation Biology* **12**(4), 749-758.
Hale, W. G. (1974). Aerial counting of waders. *Ibis* **116**, 412.
Halse, S. A., Pearson, G. B., Jaensch, R. P., Kulmoi, P., Gregory, P., Kay, W. R., and Storey, A. W. (1996). Waterbird Surveys of the Middle Fly River Floodplain, Papua New Guinea. *Wildlife Research* **23**, 557-569.
Hamer, A. J., Lane, S. J., and Mahony, M. J. (2002). Management of freshwater wetlands for the endangered green and golden bell frog (*Litoria aurea*): roles of habitat determinants and space. *Biological Conservation* **106**(3), 413-424.
Handel, C. M., and Gill, R. E. J. (1992). Roosting Behaviour of Premigratory Dunlins (*Calidris alpina*). *The Auk* **109**(1), 57-72.
Harding, J., Harding, S., and Driscoll, P. (1999). Empire Point Roost: a purpose built roost site for waders. *The Stilt* **34**, 46-49.
Harris, K. (1994a). Population Monitoring Counts. *The Stilt* **25**, 12-19.
Harris, K. (1994b). Population Monitoring Counts. *The Stilt* **24**, 11.
Harris, K. (1995a). Population Monitoring Counts: Winter 1994. *The Stilt* **26**, 21-22.
Harris, K. (1995b). Population Monitoring Counts: Summer 1995. *The Stilt* **27**, 27-29.
Harris, K. (1996). Population monitoring counts: a report on the winter 1995 and summer 1996 counts. *The Stilt* **29**, 43-47.
Harris, K. (1997). Report on Population Monitoring Counts. *The Stilt* **31**, 56-62.
Harris, K. (1999). Report on Population Monitoring Counts. *The Stilt* **34**, 51-54.
Harris, K. (2000). Report on Population Monitoring Counts. *The Stilt* **36**, 45-46.
Havens, K. J., Varnell, L. M., and Watts, B. D. (2002). Maturation of a constructed tidal marsh relative to two natural reference tidal marshes over 12 years. *Ecological Engineering* **18**, 305-315.
Hazelton, P. A., and Murphy, B. W. (1992). 'What Do All the Numbers Mean? A Guide for the Interpretation of Soil Test Results'. (NSW Department of Conservation and Land Management: Sydney).
HBOC. (1979). Interesting Observations of Bird Species in the Hunter Region During 1978. *Hunter Natural History* **11**, 24-25.
HCRCMA (2009). Hunter Central Rivers Catchment Management Authority Hexham Swamp Rehabilitation Project. Available at: http://www.hcr.cma.nsw.gov.au/hexham_swamp.php3 (accessed 25 February 2009).
Hedge, P., and Kriwoken, L. K. (2000). Evidence for effects of *Spartina anglica* invasion on benthic macrofauna in Little Swanport estuary, Tasmania. *Austral Ecology* **25**, 150-159.
Herbert, C. (2007). Distribution, Abundance and Status of Birds in the Hunter Estuary. *Hunter Bird Observers Club Special Report*, No. 4.
Hewish, M. (1986). The AWSG Population Monitoring Project: report on the Summer 1986 Wader Count. *The Stilt* **9**, 21-29.
Hewish, M. (1987a). Report of the Winter 1986 Population Monitoring Count: a bumper year for Red Knots and Grey Plovers. *The Stilt* **11**, 18-22.
Hewish, M. (1987b). The Summer 1987 Population Monitoring Count: rarities and the wader counts. *The Stilt* **11**, 23-31.
Hewish, M. (1988). The Winter 1987 Population Monitoring Count: Lesser Golden Plovers over-wintering in Australia. *The Stilt* **12**, 41-46.
Hewish, M. (1989a). The Winter 1988 Population Monitoring Count. *The Stilt* **15**, 13-15.

Hewish, M. (1989b). The Summer 1988 Population Monitoring Count: Banded Stilts at monitored sites in south-eastern Australia. *The Stilt* **14**, 14-20.

Hewish, M. (1990a). The Summer 1989 Population Monitoring Count: increasing numbers of Bar-tailed Godwits at monitored sites in eastern Australia, 1982-1989. *The Stilt* **16**, 23-35.

Hewish, M. (1990b). The Winter 1989 Population Monitoring Count: Pied and Sooty Oystercatchers at monitored sites. *The Stilt* **17**, 5-16.

Hewish, M. (1992). The 1990 Summer and Winter Population Monitoring Counts. *The Stilt* **20**, 33-37.

Higgins, P. J., and Davies, S. J. J. F. (1996). 'Handbook of Australian, New Zealand and Antarctic Birds'. Volume 3: Snipe to Pigeons. (Oxford University Press Australia: Melbourne).

Hintze, J. (2008). PASS 2008. NCSS, LLC: Kaysville, Utah.

Hobson, K. A. (1999). Tracing origins and migration of wildlife using stable isotopes: a review. *Oecologia* **120**, 314-326.

Hoenig, J. M., Robson, D. S., Pollock, C. M., and Jones, K. H. (1993). Scheduling counts in the instantaneous and progressive count methods for estimating sport fishing effort. *North American Journal of Fisheries Management* **13**, 723-736.

Holmes, G. (1970). The Birds of the Hunter River Estuary. *Hunter Natural History* **2**(1), 13-18.

Hotker, H. (2000). When do Dunlins spend high tide in flight? *Waterbirds* **23**(3), 482-485.

Howe, A. (2008). Hydrodynamics, Geomorphology and Vegetation of Estuarine Wetlands in the Hunter, Australia: Implications for Migratory Shorebird High Tide Roost Availability. Ph.D. Thesis, Civil, Surveying and Environmental Engineering, University of Newcastle, Australia.

Howe, A. J., Rodríguez, J. F., Spencer, J. A., MacFarlane, G. R., and Saintilan, N. (in press). Response of estuarine wetlands to reinstatement of tidal flows. *Marine and Freshwater Research*,

Howe, M. A., and Geissler, P. H. (1989). Population Trends of North American Shorebirds Based on International Shorebird Survey. *Biological Conservation* **49**, 185-199.

Hurt, A., Hansbro, P., Selleck, P., Olsen, B., Minton, C., Hampson, A., and Barr, I. (2006). Isolation of avian influenza viruses from two different transhemispheric migratory shorebird species in Australia. *Archives of Virology* **151**(11), 2301-2309.

Hutchings, P. (1977). A Survey of Fullerton Cove. (Australian Littoral Society: Sydney).

Hutchings, P. (1983). The Wetlands of Fullerton Cove, Hunter River, New South Wales. *Wetlands (Australia)* **3**(1), 12-21.

Hutchinson, A., and Morris, A. K. (2003). Rehabilitation of Waterbird Habitat on Ash Island. Final report prepared for the Environmental Trust Environmental Protection Authority. (Birds Australia: Sydney).

HWC (2007). Why Tillegra Now? Background Paper. (NSW Hunter Water Corporation: Newcastle).

IPCC (2007). Climate Change 2007: Synthesis Report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC: Geneva, Switzerland.

Jessop, A. (2003). Gut analysis of five wader species collected from the NW of Western Australia. *The Stilt* **43**, 3-13.

Jessop, R., and Collins, P. (1999). Observations of feeding and roosting behaviour of Little Curlew *Numenius minutus* overwintering in the North-west of Australia. *The Stilt* **35**, 62-63.

Johnson, D. H. (1980). The Comparison of Usage and Availability Measurements for Evaluating Resource Preference. *Ecology* **61**(1), 65-71.

Jones, A. R., Watson-Russell, C. J., and Murray, A. (1986). Spatial Patterns in the Macrobenthic Communities of the Hawkesbury Estuary, New South Wales. *Australian Journal of Marine and Freshwater Research* **37**, 521-543.

Jones, E. L. (1985). The Influence of Tide Height on Waders' Choice of Roost Sites. *Victorian Wader Study Group Bulletin* **9**, 2-6.

Kalejta, B. (1992). Time budgets and predatory impact on waders at the Berg River Estuary, South Africa. *Ardea* **80**, 327-342.

Kennish, M. J. (2002). Environmental threats and environmental future of estuaries. *Environmental Conservation* **29**(1), 78-107.

Kent, A. C., and Day, R. W. (1983). Population dynamics of an infaunal polychaete: the effect of predators and an adult-recruit interaction. *Journal of Experimental Marine Biology and Ecology* **73**, 185-203.

Kingsford, R. T. (1990). The effect of human activities on shorebirds, seabirds and waterbirds of Comerong Island at the mouth of the Shoalhaven River. *Wetlands (Australia)* **9**, 7-12.

Kingsford, R. T., and Porter, J. L. (1993). Waterbirds of Lake Eyre, Australia. *Biological Conservation* **65**, 141-151.

Kingsford, R. T., and Porter, J. L. (1994). Waterbirds on an adjacent freshwater lake and salt lake in arid Australia. *Biological Conservation* **69**(2), 219-228.

Kingsford, R. T., and Ferster Levy, R. (1997). Changes to the Hunter Estuary, 1801-1996, and their implications for migratory wading birds and other waterbirds. Unpublished report to Environment Australia. (NSW National Parks and Wildlife Service: Hurstville, Sydney).

Kingsford, R. T., Ferster Levy, R., Geering, D., Davis, S. T., and Davis, J. S. E. (1998). Kooragang Wetland Rehabilitation Project: Rehabilitating estuarine habitat on Kooragang Island for waterbirds, including migratory wading birds - (May 1994 - May 1997). (NSW National Parks and Wildlife Service: Hurstville, Sydney).

Kingsford, R. T. (1999). Aerial survey of waterbirds on wetlands as a measure of river and floodplain health. *Freshwater Biology* **41**, 425-438.

Kingsford, R. T., Wong, P. S., Braithwaite, L. W., and Maher, M. T. (1999). Waterbird abundance in eastern Australia, 1983-92. *Wildlife Research* **26**, 351-366.

Kingsford, R. T. (2000). Review: Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Austral Ecology* **25**, 109-127.

Kingsford, R. T., Brandis, K., Thomas, R., Crighton, P., Knowles, E., and Gale, E. (2004). Classifying landform at broad spatial scales: the distribution and conservation of wetlands in New South Wales, Australia. *Marine and Freshwater Research* **55**, 17-31.

Kingsford, R. T., and Thomas, R. F. (2004). Destruction of Wetlands and Waterbird Populations by Dams and Irrigation on the Murrumbidgee River in Arid Australia. *Environmental Management* **34**, 383-396.

Kingsford, R. T., and Porter, J. L. (2006). Waterbirds and wetlands across Eastern Australia. (Online) Technical report prepared for the Department of the Environment and Heritage, Canberra. University of New South Wales and NSW National Parks and Wildlife Service: Sydney. Available at: http://www.environment.gov.au/soe/2006/publications/technical/waterbirds/pubs/waterbirds.pdf (accessed 28 January 2009).

Kingsford, R. T., and Porter, J. L. (2009). Monitoring waterbird populations with aerial surveys - what have we learnt. *Wildlife Research* **36**(1), 29-40.

Kwak, T. J., and Zedler, J. B. (1997). Food web analysis of southern California coastal wetlands using multiple stable isotopes. *Oecologia* **110**(2), 262-277.

Laegdsgaard, P. (2006). Ecology, disturbance and restoration of coastal saltmarsh in Australia: a review. *Wetlands Ecology and Management* **14**, 379-399.

Landys-Ciannelli, M. M., Piersma, T., and Jukema, J. (2003). Strategic size changes of internal organs and muscle tissue in the Bar-tailed Godwit during fat storage on a spring stopover site. *Functional Ecology* **17**(2), 151-159.

Lane, B. (1987). 'Shorebirds in Australia'. (Nelson Publishers: Melbourne).

Lawler, W. (1994). Draft Management Manual for Migratory Shorebird Feeding Habitat in New South Wales Estuaries. (NSW National Parks and Wildlife Service: Hurstville, Sydney).

Lawler, W. (1995). Wader Roost Construction In Moreton Bay: A Feasibility Study into the Construction of Migratory Wader (Shorebird) High Tide Roosts in Moreton Bay, Qld, using Raby Bay as a Case Study. A report for the Queensland Department of Natural Resources. (Queensland Wader Study Group: Brisbane).

Lawler, W. (1996). Guidelines for Management of Migratory Shorebird Habitat in Southern East Coast Estuaries, Australia. M.Sc. Thesis, Department of Ecosystem Management, University of New England, Armidale.

Lawrence, A. J., and Soame, J. M. (2004). The effects of climate change on the reproduction of coastal invertebrates. *Ibis* **146**(1), 29-39.

Lee, S. Y., Dunn, R. J. K., Young, R. A., Connolly, R. M., Dale, P. E. R., Dehayr, R., Lemckert, C. J., McKinnon, S., Powell, B., Teasdale, P. R., and Welsh, D. T. (2006). Impact of urbanization on coastal wetland structure and function. *Austral Ecology* **31**, 149-163.

Lendrem, D. (1983). Sleeping and vigilance in birds. I. Field Observations of the Mallard (*Anas platyrhynchos*). *Animal Behaviour* **31**, 532-538.

Lendrem, D. (1984). Sleeping and vigilance in birds, II. An experimental study of the Barbary Dove (*Streptopelia risoria*). *Animal Behaviour* **32**, 243-248.

Lindegarth, M., and Chapman, M. G. (2001). Testing hypotheses about management to enhance habitat for feeding birds in a freshwater wetland. *Journal of Environmental Management* **62**, 375-388.

Lindsey, T. (1979). New South Wales Bird Report for 1978. *Australian Birds* **14**(1), 1-22.

Lindsey, T. (1980). New South Wales Bird Report for 1979. *Australian Birds* **15**(2), 17-26.

Lindsey, T. (1981). New South Wales Bird Report for 1980. *Australian Birds* **16**(1), 1-23.

Lindsey, T. (1982). New South Wales Bird Report for 1981. *Australian Birds* **17**(1), 1-26.

Lindsey, T. (1984). New South Wales Bird Report for 1982. *Australian Birds* **18**(3), 37-69.

Lindsey, T. (1985). New South Wales Bird Report for 1983. *Australian Birds* **19**(4), 65-100.

Lindsey, T. (1986). New South Wales Bird Report for 1984. *Australian Birds* **20**(4), 97-132.

Lindström, A., and Agrell, J. (1999). Global change and possible effects on the migration and reproduction of Arctic-breeding waders. *Ecological Bulletins* **47**, 145-159.

Luis, A., Goss-Custard, J. D., and Moreira, M. H. (2001). A method for assessing the quality of roosts used by waders during high tide. *Wader Study Group Bulletin* **96**, 71-73.

Luis, A., Goss-Custard, J. D., and Moreira, M. H. (2002). The feeding strategy of the dunlin (*Calidris alpina* L.) in artificial and non-artificial habitats at Ria de Aveiro, Portugal. *Hydrobiologia* **475/476**, 335-343.

Ma, Z. J., Li, B., Zhao, B., Jing, K., Tang, S. M., and Chen, J. K. (2004). Are artificial wetlands good alternatives to natural wetlands for waterbirds? A case study on Chongming Island, China. *Biodiversity and Conservation* **13**(2), 333-350.

MacDonald, T. A. (2001). Investigating the estuarine wetlands of the lower Hunter River: rehabilitation potential of tidal reinstatement following degradation caused by tidal restriction. Ph.D. Thesis, University of Newcastle, Australia.

MacFarlane, G. (2002). Non-destructive sampling techniques for the rapid assessment of populations parameters in estuarine shore crabs. *Wetlands (Australia)* **20**(2), 49-54.

Marchant, S., and Higgins, P. J. (1993). 'Handbook of Australian, New Zealand and Antarctic Birds'. Volume 2: Raptors to Lapwings. (Oxford University Press: Melbourne).

Martindale, J. (1982). A Study of Wading Birds in Corner Inlet. Technical Report Series. Report No. 4. (Arthur Rylah Institute for Environmental Research, Ministry for Conservation and Wildlife Division: Victoria).

Masero, J., A, Perez-Gonzalez, M., Bassadre, M., and Otero-Saavedra, M. (1999). Food supply for waders (Aves: Charadrii) in an estuarine area in the Bay of Cadiz (SW Iberian Peninsula). *Acta Oecologica-International Journal of Ecology* **20**(4), 429-434.

Masero, J. A., Perez-Hurtado, A., Castro, M., and Arroyo, G. M. (2000). Complementary use of intertidal mudflats and adjacent salinas by foraging waders. *Ardea* **88**(2), 177-191.

Masero, J. A., and Perez-Hurtado, A. (2001). Importance of the supratidal habitats for maintaining overwintering shorebird populations: how redshanks use tidal mudflats and adjacent saltworks in southern Europe. *The Condor* **103**, 21-30.

Masero, J. A. (2003). Assessing alternative anthropogenic habitats for conserving waterbirds: salinas as buffer areas against the impact of natural habitat loss for shorebirds. *Biodiversity and Conservation* **12**(6), 1157-1173.

Mazumder, D., and Saintilan, N. (2003). A comparison of sampling techniques in the assessment of burrowing crab abundance in saltmarsh and mangrove environments. *Wetlands (Australia)* **21**(1), 1-15.

Mazumder, D., Saintilan, N., and Williams, R. (2006). Fish Assemblages in Three Tidal Saltmarsh and Mangrove Flats in Temperate NSW, Australia: A Comparison Based on Species Diversity and Abundance. *Wetlands Ecology and Management* **14**(3), 201-209.

McKinstry, M. C., and Anderson, S. H. (2002). Creating wetlands for waterfowl in Wyoming. *Ecological Engineering* **18**, 293-304.

Meire, P. M., and Kuyen, E. (1984). Relations between the distribution of waders and the intertidal benthic fauna of the Oosterschelde, Netherlands. In 'Coastal Waders and Wildfowl in Winter'. (Ed. P. R. Evans, J. D. Goss-Custard, and W. G. Hale). pp. 57-68. (Cambridge University Press: Cambridge, UK).

Meltofte, H. (2001). Wader population censuses in the Arctic: Getting the timing right. *Arctic* **54**(4), 367-376.

Melvin, S. L., and Webb, J. W. (1998). Differences in the avian communities of natural and created *Spartina alterniflora* salt marshes. *Wetlands* **18**(1), 59-69.

Metcalfe, N. B. (1984a). The effects of habitat on the vigilance of shorebirds: is visibility important? *Animal Behaviour* **32**, 981-985.

Metcalfe, N. B. (1984b). The effects of mixed-species flocking on the vigilance of shorebirds: Who do they trust? *Animal Behaviour* **32**, 986-993.

MHL (2003). Hunter Estuary Processes Study. Report No. MHL 1095 to the NSW Department of Commerce. (Manly Hydraulics Laboratory: Manly Vale, NSW).

Minton, C., Jessop, R., and Collins, P. (2003). Variations in apparent annual breeding success of red-necked stints and curlew sandpipers between 1991 and 2001. *The Stilt* **43**, 30-33.

Minton, C. (2005). What Have We Learned from Banding and Flagging Waders in Australia? In 'Status and Conservation of Shorebirds in the East Asian-Australasian Flyway; Proceedings of the Australasian Shorebird Conference 13-15 December 2003, Canberra,

Australia.' (Ed. P. Straw). pp. 116-142. (Australasian Wader Studies Group and Wetlands International - Oceania.).
Minton, C., Jessop, R., Collins, P., and Hassell, C. (2005). Juvenile percentages of migratory waders in the 2004/05 Australian summer. *Stilt* **47**, 10-14.
Minton, C., Wahl, J., Jessop, R., Hassell, C., Collins, P., and Gibbs, H. (2006). Migration Routes of Waders which spend the Non-Breeding Season in Australia. *Stilt* **50**, 135-157.
Mitchell, J. R., Moser, M. E., and Kirby, J. S. (1988). Declines in midwinter counts of waders roosting on the Dee estuary. *Bird Study* **35**, 191-198.
Mitsch, W. J., and Gosselink, J. G. (2000). 'Wetlands'. (John Wiley & Sons Inc: New York).
Mizutani, H., Fukuda, M., Kabaya, Y., and Wada, E. (1990). Carbon Isotope Ratio of Feathers Reveals Feeding Behaviour of Cormorants. *The Auk* **107**, 400-403.
Moores, N. (2006). South Korea's Shorebirds: A Review of Abundance, Distribution, Threats and Conservation Status. *Stilt* **50**, 62-72.
Moores, N., Battley, P., Rogers, D., Park, M.-N., Sung, H.-C., van de Kam, J., and Gosbell, K. (2006). The Saemangeum Shorebird Monitoring Program, 2006 report. (Birds Korea - Australasian Wader Study Group).
Moores, N., Rogers, D., Koh, C.-H., Ju, Y.-K., Kim, R.-H., and Park, M.-N. (2007). The 2007 Saemangeum Shorebird Monitoring Program Report. (Birds Korea-Australasian Wader Study Group).
Moores, N., Rogers, D., Kim, R.-H., Hassel, C., Gosbell, K., Kim, S.-A., and Park, M.-N. (2008). The 2006-2008 Saemangeum Shorebird Monitoring Program Report. (Birds Korea: South Korea).
Morgan, T. (2008). Toxicity of the larvicide, *Bacillus thuringiensis israelensis* to the brackish water chironomid species, *Kiefferulus longilobus* (Diptera: Chironomidae). Hons. Thesis, Faculty of Sciences, University of Technology Sydney, Australia.
Morris, A. K. (1975). The Birds of Gosford, Wyong and Newcastle (County of Northumberland). *Australian Birds* **9**(3), 37-59.
Morris, A. K. (1983). Review of the Policies, Priorities and Programmes for Nature Conservation in New South Wales: Waders, Gulls and Terns. Paper No. 52. (NSW National Parks and Wildlife Service: Hurstville).
Morris, A. K., and Burton, A. (1992). New South Wales Annual Bird Report - 1989. *Australian Birds* **26**(2), 41-70.
Morris, A. K., and Burton, A. (1993). New South Wales Annual Bird Report 1991. *Australian Birds* **27**(2), 29-76.
Morris, A. K., and Burton, A. (1994). New South Wales Annual Bird Report 1992. *Australian Birds* **27**(4), 97-139.
Morris, A. K., and Burton, A. (1995). New South Wales Annual Bird Report 1993. *Australian Birds* **28**(4), 81-128.
Morris, A. K., and Burton, A. (1996). New South Wales Annual Bird Report 1994. *Australian Birds* **29**(4), 63-112.
Morris, A. K., and Burton, A. (1997). New South Wales Annual Bird Report 1995. *Australian Birds* **30**(4), 81-149.
Morris, A. K., and Burton, A. (1999). New South Wales Annual Bird Report 1996. *Australian Birds* **31**(4), 86-129.
Morris, A. K. (2000). New South Wales Annual Bird Report 1997. *Australian Birds* **32**(1), 1-64.
Morris, A. K. (2001). New South Wales Annual Bird Report 1998. *Australian Birds* **32**(3), 101-166.
Morris, A. K. (2002a). New South Wales Annual Bird Report 2000. *Australian Birds* **33**(1), 1-75.

Morris, A. K. (2002b). New South Wales Annual Bird Report 1999. *Australian Birds* **32**(4), 173-245.

Morris, A. K. (2003). New South Wales Annual Bird Report 2001. *Australian Birds* **33**(2), 85-160.

Morris, A. K. (2004). New South Wales Annual Bird Report 2002. *Australian Birds* **33**(3), 165-242.

Morrisey, D. J., Underwood, A. J., Howitt, L., and Stark, J. S. (1992). Temporal variation in soft-sediment benthos. *Journal of Experimental Marine Biology and Ecology* **164**(2), 233-245.

Morrison, R. I. G., Ross, R. K., and Niles, L. J. (2004). Declines in wintering populations of Red Knots in southern South America. *Condor* **106**(1), 60-70.

Morton, S. R., Brennan, K. G., and Armstong, M. D. (1990). Distribution and Abundance of Ducks in the Alligator Rivers Region, Northern Territory. *Australian Wildlife Research* **17**, 573-590.

Morton, S. R., Brennan, K. G., and Armstong, M. D. (1993). Distribution and Abundance of Grebes, Pelicans, Darters, Cormorants, Rails and Terns in the Alligator Rivers Region, Northern Territory. *Wildlife Research* **20**, 203-217.

Moser, M., Prentice, C., and Frazier, S. (1996). A Global Overview of Wetland Loss and Degradation. Proceedings from the 6th Meeting of the Conference of the Contracting Parties in Brisbane, Australia, March 1996. (Ramsar Convention Bureau: Gland, Switzerland).

Moser, M. E. (1988). Limits to the numbers of grey plovers *Pluvialis squatarola* wintering on British estuaries: an analysis of long-term population trends. *Journal of Applied Ecology* **25**, 473-485.

Moss, J. (1983). Investigation of Natural Areas of Kooragang Island. Report for the NSW Department of Environment and Planning. (C. D. Field and Associates: Sydney).

Mouritsen, K. N. (1994). Day and night feeding in Dunlins *Calidris alpina*: choice of habitat, foraging technique and prey. *Journal of Avian Biology* **25**, 55-62.

Moy, L. D., and Levin, L. A. (1991). Are Spartina Marshes a Replaceable Resource? A Functional Approach to Evaluation of Marsh Creation Efforts. *Estuaries* **14**(1), 1-16.

Munro, D. C., and Touron, H. (1997). The estimation of marshland degradation in southern Iraq using multitemporal Landsat TM images. *International Journal of Remote Sensing.* **18**(7), 1597-1606.

Naismith, L. (1992). Results of the 1991 Summer Population Monitoring Counts. *The Stilt* **21**, 6-7.

Nebel, S., Piersma, T., Van Gils, J., Dekinga, A., and Spaans, B. (2000). Length of stopover, fuel storage and a sex-bias in the occurrence of red knots *Calidris c. canutus* and *C-c. islandica* in the Wadden Sea during southward migration. *Ardea* **88**(2), 165-176.

Nebel, S. (2007). Differential migration of shorebirds in the East-Asian Australasian Flyway. *Emu* **107**(1), 14-18.

Nebel, S., Porter, J. L., and Kingsford, R. T. (2008). Long-term trends of shorebird populations in eastern Australia and impacts of freshwater extraction. *Biological Conservation* **141**, 971-980.

Neckles, H. A., Dionne, M., Burdick, D. M., Roman, C. T., Buchsbaum, R., and Hutchins, E. (2002). A Monitoring Protocol to Assess Tidal Restoration of Salt Marshes on Local and Regional Scales. *Restoration Ecology* **10**(3), 556-563.

Neira, C., Grosholz, E., D, Levin, L., A, and Blake, R. (2006). Mechanisms generating modification of benthos following tidal flat invasion by a Spartina hybrid. *Ecological Applications* **16**(4), 1391-1404.

Nelson, P. (2006). Ecological Studies in the Restoration of Estuarine Wetland Habitats in Eastern Australia. Ph.D. Thesis, University of Newcastle, Australia.

Nicholls, R., Hoozemans, F. M. J., and Marchand, M. (1999). Increasing flood risk and wetland losses due to global sea-level rise: regional and global analyses. *Global Environmental Change* **9**(1), 69-87.

NLWRA. (2002). Australian Catchment, River and Estuary Assessment 2002. (National Land and Water Resources Audit: Canberra).

Norris, K. (2000). The conservation and management of saltmarshes for birds. In 'British Saltmarshes'. (Ed. B. R. Sherwood, B. G. Gardiner, and T. Harris). pp. 341-357. (Forrest Text: UK).

NPC (2009). Port development: future projects. Newcastle Port Corporation. Available at: http://www.newportcorp.com/site/index.cfm?display=111650 (accessed 28 Jan 2009).

NPWS (1998a). Kooragang Nature Reserve and Hexham Swamp Nature Reserve Plan of Management. (NSW National Parks and Wildlife Service: Hunter District).

NPWS (1998b). Seven Mile Beach National Park and Comerong Island Nature Reserve Plan of Management. (NSW National Parks and Wildlife Service: Nowra).

Ntiamoa-Baidu, Y., Piersma, T., Wiersma, P., Poot, M., Battley, P. F., and Gordon, C. (1998). Water depth selection, daily feeding routines and diets of waterbirds in coastal lagoons in Ghana. *Ibis* **140**, 89-103.

Olsen, P., Weston, M., Cunningham, R., and Silcocks, A. (2003). The State of Australia's Birds 2003. Supplement to Wingspan Volume 13 (4). (Birds Australia: Melbourne).

Olsen, P., and Weston, M. (2004). The State of Australia's Birds 2004: Water, Wetlands and Birds. Supplement to Wingspan Volume 14 (4). (Birds Australia: Melbourne).

Outhred, R. K., and Buckney, R. T. (1983). The vegetation of Kooragang Island, New South Wales. *Wetlands: Journal of the Coast and Wetland Society* **3**(2), 58 - 70.

Page, G., and Whitacre, D. F. (1975). Raptor predation on wintering shorebirds. *The Condor* **77**, 73-83.

Parsons, K. C. (2002). Integrated management of waterbird habitats at impounded wetlands in Delaware Bay, USA. *Waterbirds* **25**, 25-41.

Paton, D., C, Ziembicki, M., Owen, P., and Heddle, C. (2000a). Disturbance distances for water birds and the management of human recreation with special reference to the Coorong region of South Australia. *The Stilt* **37**, 46.

Paton, D. C., Ziembicki, M., Owen, P., Hill, B., and Bailey, C. (2000b). Distribution and abundance of migratory waders and their food in the Estuarine areas of the Murray mouth and patterns in the composition of sediments. *The Stilt* **37**, 47.

Paton, D. C., and Rogers, D. J. (2006). The Coorong, Lake Alexandrina and Lake Albert. Waterbird research in the Coorong Region, South Australia: past, present, future. *The Stilt* **50**, 254-256.

Paton, P. A. (1984). Diet of Waders in the Coorong, S.A. *The Stilt* **5**, 27.

Pennycuick, C. J., and Battley, P. F. (2003). Burning the engine: a time-marching computation of fat and protein consumption in a 5420-km non-stop flight by great knots, *Calidris tenuirostris*. *Oikos* **103**, 323-332.

Pfister, C., Harrington, B. A., and Lavine, M. (1992). The impact of human disturbance on shorebirds at a migration staging area. *Biological Conservation* **60**, 115-126.

Phillips, W., and Muller, K. (2006). Ecological Character of the Coorong, Lake Alexandrina and Albert Wetland of International Importance. (South Australian Department of Environment and Heritage: Adelaide).

Pienkowski, M. W. (1983). Surface activity of some intertidal invertebrates in relation to temperature and the foraging behaviour of their shorebird predators. *Marine Ecology Progress Series* **11**, 141-150.

Piersma, T., Gudmundsson, G. A., and Lilliendahl, K. (1999). Rapid Changes in the Size of Different Functional Organ and Muscle Groups during Refuelling in a Long-Distance Migrating Shorebird. *Physiological and Biochemical Zoology* **72**(4), 405-415.

Pontee, N. I. (2003). Designing sustainable estuarine intertidal habitats. *Proceedings of the Institution of Civil Engineers: Engineering Sustainability* **156**(ES3), 157-167.

Poore, G. C. B., Corrick, A. H., and Norman, F. I. (1979). Food of three waders at Lake Reeve, Victoria. *Emu* **79**, 228-229.

Porter, J. L., Kingsford, R. T., and Hunter, S. (2006). Aerial Surveys of Wetland Birds in Eastern Australia - October 2003-2005. Occasional Paper No. 37. (NSW Department of Environment and Conservation: Hurstville).

Pressey, R. L., and Middleton, M. J. (1982). Impacts of Flood Mitigation works on Coastal Wetlands in New South Wales. *Wetlands (Australia)* **2**(1), 27-44.

PRIMER. (2002). PRIMER 5 for Windows. Version 5.2.9. PRIMER-E Ltd.

Prys-Jones, R. P., Underhill, L., and Waters, R. J. (1994). Index numbers for waterbird populations. II. Coastal wintering waders in the United Kingdom, 1970/71-1990/91. *Journal of Applied Ecology* **31**, 481-492.

Puttick, G. M. (1979). Foraging behaviour and activity budgets of Curlew Sandpipers. *Ardea* **67**, 111-122.

Puttick, G. M. (1980). Energy Budgets of Curlew Sandpipers at Langebaan Lagoon, South Africa. *Estuarine and Coastal Marine Science* **11**, 207-215.

Quinn, G., and Keough, M. (2002). 'Experimental design and data analysis for biologists'. (Cambridge University Press: UK).

Ramsar (2008). The Annotated Ramsar Site List. Available at: http://www.ramsar.org/index_list.htm (accessed 18 July 2008).

Ramsar (2009). The Ramsar Convention on Wetlands Criteria for Identifying Wetlands of International Importance. Available at: http://www.ramsar.org/key_criteria.htm (accessed 24 February 2009).

Rappoldt, C., Kersten, M., and Smit, C. (1985). Errors in large-scale shorebird counts. *Ardea* **73**, 13-24.

Rattenborg, N. C., Lima, S. L., and Amlaner, C. J. (1999). Half-awake to the risk of predation. *Nature* **397**, 397-398.

Recher, H. F. (1966). Some Aspects of the Ecology of Migrant Shorebirds. *Ecology* **47**(3), 393-407.

Reeder, W. G. (1951). Stomach analysis of a group of shorebirds. *Condor* **53**, 43-45.

Rehfisch, M. M. (1994). Man-made lagoons and how their attractiveness to waders might be increased by manipulating the biomass of an insect benthos. *Journal of Applied Ecology* **31**, 383-401.

Rehfisch, M. M., Clark, N. A., Langston, R. H. W., and Greenwood, J. J. D. (1996). A guide to the provision of refuges for waders: an analysis of 30 years of ringing data from the Wash, England. *Journal of Applied Ecology* **33**, 673-687.

Reid, T., and Park, P. (2003). Continuing decline of Eastern Curlew, *Numenius madagascariensis*, in Tasmania. *Emu* **103**, 279-283.

Reigen, A. C. (1999). Movements of banded Arctic waders to and from New Zealand. *Notornis: Ornithological Society of New Zealand* **46**, 123-142.

Ribeiro, P. D., Iribarne, O. O., Navarro, D., and Jaureguy, L. (2004). Environmental heterogeneity, spatial segregation of prey, and the utilization of southwest Atlantic mudflats by migratory shorebirds. *Ibis* **146**(4), 672-682.

Richardson, A. L. (2004). Ecological Niche of the Bar-tailed Godwits (*Limosa lapponica*) in the Hunter River Estuary: Behavioural use of different habitats. Hons. Thesis, Biological Sciences, University of Newcastle, Australia.

RIS. (1994). Information Sheet on Ramsar Wetlands (RIS): Olango Island, Philippines. (Wetlands International Ramsar Information website: http://www.wetlands.org/rsis/ (accessed 20 July 2008)).

RIS (1999a). Information Sheet on Ramsar Wetlands (RIS): Western Port, Victoria. Wetlands International Ramsar Information. Available at: http://www.wetlands.org/rsis/ (accessed 20 July 2008).

RIS (1999b). Information Sheet on Ramsar Wetlands: Corner Inlet, Victoria. Wetlands International Ramsar Information. Available at: http://www.wetlands.org/rsis/ (accessed 24 July 2008).

RIS (2001a). Information Sheet on Ramsar Wetlands (RIS): Krabi Estuary, Thailand. Wetlands International Ramsar Information. Available at: http://www.wetlands.org/rsis/ (accessed 20 July 2008).

RIS (2001b). Information Sheet on Ramsar Wetlands: Chongming Dongtan Nature Reserve, Shanghai. Wetlands International Ramsar Information. Available at: http://www.wetlands.org/rsis/ (accessed 20 July 2008).

RIS (2002). Information Sheet on Ramsar Wetlands (RIS): Hunter Estuary Wetlands. Wetlands International Ramsar Information. Available at: http://www.wetlands.org/rsis/ (accessed 20 July 2008).

RIS (2005). Wasur National Park, Indonesia. Information Sheet for Ramsar Wetlands. Wetlands International Ramsar Information. Available at: http://www.wetlands.org/rsis/ (accessed 20 July 2008).

Roberts, G. (1996). Why individual vigilance declines as group size increases. *Animal Behaviour* **51**, 1077-1086.

Robertson, A. I., and Duke, N. C. D. (1987). Mangroves as nursery sites: comparisons of the abundance and species composition of fish and crustaceans in mangroves and other nearshore habitats in tropical Australia. *Marine Biology* **96**(2), 1432-1793.

Robinson, K., and Gibbs, P. (1982). 'A field guide to the common shelled molluscs of New South Wales estuaries'. (Coast and Wetland Society: Sydney).

Rogers, A. E. F. (1972). NSW Bird Report for 1971. *NSW Birds* **6**(6), 77-99.

Rogers, A. E. F. (1973). NSW Bird Report for 1972. *NSW Birds* **7**(6), 89-108.

Rogers, A. E. F. (1974). NSW Bird Report for 1973. *NSW Birds* **8**(6), 97-118.

Rogers, A. E. F. (1975). NSW Bird report for 1974. *Australian Birds* **9**(4), 77-97.

Rogers, A. E. F. (1976). NSW Bird Report for 1975. *Australian Birds* **10**(4), 61-84.

Rogers, A. E. F. (1977). NSW Bird Report for 1976. *Australian Birds* **11**(4), 81-104.

Rogers, A. E. F., and Lindsey, T. R. (1978). NSW Bird Report for 1977. *Australian Birds* **13**(1), 1-24.

Rogers, D. I. (1999). What determines shorebird feeding distribution in Roebuck Bay? In 'Intertidal sediments and benthic animals of Roebuck Bay, Western Australia.'. (Ed. M. Pepping, T. Piersma, G. Pearson, and M. Lavaleye). pp. 145-174. (Netherlands Institute

for Sea Research (NIOZ). Western Australia Department of Conservation and Land Management (CALM). Curtin University of Technology, Perth, W.A.).

Rogers, D. I. (2003). High-tide roost choice by coastal waders. *Wader Study Group Bulletin* **100**, 73-79.

Rogers, D. I. (2006). Hidden Costs: Challenges Faced by Migratory Shorebirds Living on Intertidal Flats. Ph.D. Thesis, Institute of Land, Water and Society, Charles Sturt University, Albury.

Rogers, D. I., Battley, P. F., Piersma, T., Van Gils, J., and Rogers, K. G. (2006a). High-tide habitat choice: insights from modelling roost selection by shorebirds around a tropical bay. *Animal Behaviour* **72**, 563-575.

Rogers, D. I., Piersma, T., and Hassell, C. J. (2006b). Roost availability may constrain shorebird distribution: Exploring the energetic costs of roosting and disturbance around a tropical bay. *Biological Conservation* **133**(2), 225-235.

Rogers, K. G., and Gosbell, K. (2006). Demographic models for Red-necked Stint and Curlew Sandpiper in Victoria. *Stilt* **50**, 205-214.

Rohweder, D. A., and Baverstock, P. R. (1996). Preliminary Investigation of Nocturnal Habitat Use by Migratory Waders (Order Charadriformes) in Northern New South Wales. *Wildlife Research* **23**, 169-184.

Rohweder, D. A. (2000). Day-night habitat use by five species of migratory shorebird in the Richmond River Estuary, northern New South Wales, Australia. Ph.D. Thesis, Southern Cross University, Lismore.

Rohweder, D. A. (2001). Nocturnal Roost Use by Migratory Waders in the Richmond River Estuary, Northern New South Wales, Australia. *The Stilt* **40**, 23-28.

Rohweder, D. A., and Lewis, B. D. (2004). Day-night foraging behaviour of banded dotterels (*Charadrius bicinctus*) in the Richmond River estuary, northern NSW, Australia. *Notornis: Ornithological Society of New Zealand* **51**(3), 141-156.

Rojas, L. M., McNeil, R., Cabana, T., and Lachapelle, P. (1999). Diurnal and nocturnal visual capabilities in shorebirds as a function of their feeding strategies. *Brain Behavior and Evolution* **53**(1), 29-43.

Rosa, A., Encarnacao, A. L., Granadeiro, J. P., and Palmeirim, J. M. (2006). High water roost selection by waders: maximizing feeding opportunities or avoiding predation? *Ibis* **148**, 88-97.

Rosa, S., Palmeirim, J. M., and Moreira, F. (2003). Factors affecting waterbird abundance and species richness in an increasingly urbanized area of the Tagus estuary in Portugal. *Waterbirds* **26**(2), 226-232.

Roy, P. S., Williams, R. J., Jones, A. R., Yassini, I., Gibbs, P. J., Coates, B., West, R. J., Scanes, P. R., Hudson, J. P., and Nichol, S. (2001). Structure and Function of South-east Australian Estuaries. *Estuarine, Coastal and Shelf Science* **53**, 351-384.

Ruiz, G. M., Carlton, J. T., Grosholz, E. D., and Hines, A. H. (1997). Global Invasions of Marine and Estuarine Habitats by Non-Indigenous Species: Mechanisms, Extent, and Consequences. *American Zoologist* **37**, 621-632.

Rupp-Armstrong, S., and Nicholls, R. J. (2007). Coastal and estuarine retreat: A comparison of the application of managed realignment in England and Germany. *Journal of Coastal Research* **23**(6), 1418-1430.

Saintilan, N., and Williams, R. (1999). Mangrove transgression into saltmarsh environments in south-east Australia. *Global Ecology and Biogeography* **8**(2), 117-124.

Saintilan, N., and Williams, R. J. (2000). Short Note: The decline of saltmarsh in southeast Australia: results of recent surveys. *Wetlands (Australia)* **18**(2), 49-54.

Saintilan, N., and Wilton, K. (2001). Changes in the distribution of mangroves and saltmarshes in Jervis Bay, Australia. *Wetlands Ecology and Management* **9**, 409-420.

Saintilan, N. (2003). Balancing shorebird habitat requirements with mangrove conservation. In 'Status and Management of Migratory Shorebirds in Sydney.' (Ed. P. Straw). pp. 15-19. (Sydney Olympic Park Authority: Sydney).

Sanchez, M. I., Green, A. J., and Alegandre, R. (2006). Shorebird predation affects density, biomass, and size distribution of benthic chironomids in salt pans: an exclosure experiment. *Journal of the North American Benthological Society* **25**(1), 9-18.

Sanders, M. D. (2000). Enhancing food supplies for waders: inconsistent effects of substratum manipulations on aquatic invertebrate biomass. *Journal of Applied Ecology* **37**(1), 66-76.

Scheiffarth, G. (2001). The diet of Bar-tailed Godwits *Limosa lapponica* in the Wadden Sea: combining visual observations and faeces analyses. *Ardea* **89**(3), 481-494.

Schekkerman, H., Meininger, P. L., and Meire, P. M. (1994). Changes in the waterbird populations of the Oosterschelde (SW Netherlands) as a result of large-scale coastal engineering works. *Hydrobiologia* **282/283**, 509-524.

Scott, D. A. (1995). A Directory of Wetlands in the Middle East. (IUCN: Gland, Switzerland and IWRB: Slimbridge, UK).

Sheater, D. A., and Burger, J. (1992). Differential Responses of Tropical Roseate Terns to Aerial Intruders Throughout the Nesting Cycle. *The Condor* **94**(3), 712-719.

Sherfy, M. H., Kirkpartrick, R. L., and Richkus, K. D. (2000). Benthos core sampling and chironomid vertical distribution: implications for assessing shorebird food availability. *Wildlife Society Bulletin* **28**(1), 124-130.

Sitters, H. P., Gonzalez, P. M., Piersma, T., Baker, A. J., and Price, D. J. (2001). Day and Night Feeding Habitat of Red Knots in Patagonia: Profitability Versus Safety? *Journal of Field Ornithology* **72**(1), 86-95.

Skagen, S. K., and Knopf, F. L. (1994). Migrating shorebirds and habitat dynamics at a prairie wetland complex. *Wilson Bulletin* **106**(1), 91-105.

Skewes, J. (2002). Report on the 2001 Population Monitoring counts. *The Stilt* **41**, 55-61.

Skewes, J. (2003). Report on Population Monitoring counts, 2002. *The Stilt* **44**, 56-62.

Skewes, J. (2004). Report on Population Monitoring counts, 2003. *The Stilt* **46**, 86-92.

Skewes, J. (2005). Report on Population Monitoring counts, 2004. *The Stilt* **48**, 54-60.

Skewes, J. (2007). Report on Population Monitoring Counts, 2005 and 2006. *The Stilt* **51-52**, 20-32.

Smith, P. (1991). The biology and management of waders (Suborder Charadrii) in NSW. NSW National Parks and Wildlife Service Species Report No. 9. (NSW NPWS: Hurstville, Sydney).

Smith, R. T., and Atkinson, K. (1975). 'Techniques in Pedology: A Handbook for Environmental and Resource Studies'. (Paul Elek (Scientific Books) Ltd: London, UK).

Spencer, J., and Howe, A. (2008). Estuarine wetland rehabilitation and ecohydraulics: the links between hydraulics, sediment, benthic invertebrates, vegetation and migratory shorebird habitat: summary of major findings and recommendations. Final Report to the Kooragang Wetland Rehabilitation Project and Energy Australia. (Australian Catholic University and the University of Newcastle: NSW).

Spencer, J., Monamy, V., and Breitfuss, M. (2009). Saltmarsh as habitat for birds and other vertebrates. In 'Australian Saltmarsh Ecology'. (Ed. N. Saintilan). pp. 149-165. (CSIRO Publishing: Collingwood, Victoria).

SPSS. (2005). SPSS for Windows 14.0. SPSS Inc., 1989-2005: Chicago, Illinois.

Spurgeon, J. (1998). The Socio-Economic Costs and Benefits of Coastal Habitat Rehabilitation and Creation. *Marine Pollution Bulletin* **37**(8-12), 373-382.

Stillman, R., and Goss-Custard, J. (2002). Seasonal changes in the response of oystercatchers *Haematopus ostralegus* to human disturbance. *Journal of Avian Biology* **33**, 358-365.

Stillman, R. A., Caldow, W. G., Goss-Custard, J. D., and Alexander, M. J. (2000). Individual variation in intake rate: the relative importance of foraging efficiency and dominance. *Journal of Animal Ecology* **69**, 484-493.

Straw, P. (1999). Hunter River Estuary Wader Habitat Investigation. Report to NSW National Parks and Wildlife Service. (Avifauna Studies: Sydney).

Straw, P. (2000). Hunter Estuary Wader Habitat Investigation Stage 2. Report to NSW National Parks and Wildlife Service. (Avifauna Studies: Sydney).

Straw, P., and Saintilan, N. (2005). Shorebird Habitat Management in Australia - the threat of mangroves. In 'Status and Conservation of Shorebirds in the East Asian-Australasian Flyway; Proceedings of the Australasian Shorebird Conference 13-15 December 2003, Canberra, Australia.' (Ed. P. Straw). pp. 87-91. (Australasian Wader Studies Group and Wetlands International - Oceania.).

Straw, P. J., Gosbell, K. R., and Minton, C. D. T. (2006). Shorebird research in the East Asian-Australasian Flyway: looking to the future. In 'Waterbirds Around the World.' (Ed. G. C. Boere, C. A. Galbraith, and D. A. Stroud). pp. 328-331. (The Stationary Office, Edinburgh, UK).

Streever, W. J., Wiseman, L., Turner, P., and Nelson, P. (1996). Short term changes in flushing of tidal creeks following culvert removal. *Wetlands (Australia)* **15**(1), 22-30.

Streever, W. J. (1997). Trends in Australian wetland rehabilitation. *Wetlands Ecology and Management* **5**(1), 5-18.

Streever, W. J., and Genders, A. J. (1997). Effect of improved tidal flushing and competitive interactions at the boundary between salt marsh and pasture. *Estuaries* **20**, 807-818.

Streever, W. J. (1998). Kooragang Wetland Rehabilitation Project: opportunities and constraints in an urban wetland rehabilitation project. *Urban Ecosystems* **2**, 205-218.

Streever, W. J., Callaghan-Perry, M., Searles, A., Stevens, T., and Svoboda, P. (1998). Public attitudes and values for wetland conservation in New South Wales, Australia. *Journal of Environmental Management* **54**, 1-14.

Stroud, D. A., Baker, A., Blanco, D. E., Davidson, N. C., Delany, S., Ganter, B., Gill, R., González, P., Haanstra, L., Morrison, R. I. G., Scott, D. A., Thorup, O., West, R., Wilson, J., and Zöckler, C. (2006). The conservation and population status of the world's waders at the turn of the millennium. In 'Waterbirds Around the World.' (Ed. G. C. Boere, C. A. Galbraith, and D. A. Stroud). pp. 643-648. (The Stationary Office: Edinburgh, UK).

Stuart, A. (1994). Hunter Region of New South Wales 1993 Annual Bird Report. No. 1. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).

Stuart, A. (1995). Hunter Region of New South Wales 1994 Annual Bird Report. No. 2. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).

Stuart, A. (1996). Hunter Region of New South Wales 1995 Annual Bird Report. No. 3. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).

Stuart, A. (1997). Hunter Region of New South Wales 1996 Annual Bird Report. No. 4. (Hunter Bird Observers Club Inc: New Lambton: Newcastle).

Stuart, A. (1998). Hunter Region of New South Wales 1997 Annual Bird Report. No. 5. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).

Stuart, A. (1999). Hunter Region of New South Wales 1998 Annual Bird Report. No. 6. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).

Stuart, A. (2000). Hunter Region of New South Wales 1999 Annual Bird Report. No. 7. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (2001). Hunter Region of New South Wales 2000 Annual Bird Report. No. 8. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (2002). Hunter Region of New South Wales 2001 Annual Bird Report. No. 9. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (2003). Hunter Region of New South Wales 2002 Annual Bird Report. No. 10. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (2004). Hunter Region of New South Wales 2003 Annual Bird Report. No. 11. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (2005). Hunter Region of New South Wales 2004 Annual Bird Report. No. 12. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (2006). Hunter Region of New South Wales 2005. Annual Bird Report. No. 13. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (2007). Hunter Region of New South Wales 2006 Annual Bird Report. No. 14. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (2008). Hunter Region of New South Wales 2007 Annual Bird Report. No. 15. (Hunter Bird Observers Club Inc: New Lambton, Newcastle).
Stuart, A. (*in prep.*). Shorebird Records of the Hunter Estuary prior to April 1999. Special Publication. (Hunter Bird Observers Club: New Lambton, Newcastle).
Svoboda, P. (1996). Kooragang Wetland Rehabilitation Project Management Plan. (Hunter Catchment Management Trust: Paterson).
Swineboard, J. (1961). Nocturnal roosts of migrating shorebirds. *Wilson Bulletin* **76**, 155-159.
Taylor, I. R., and Bester, A. (1999). The response of foraging waders to human recreation disturbance at Rhyll, Phillip Island, Victoria. *The Stilt* **35**, 67.
Taylor, I. R., Taylor, S. G., and Larmour, G. N. (1999). Sex-related differences in the foraging behaviour of Bar-tailed Godwits *Limosa lapponica* in New South Wales, Australia. *The Stilt* **35**, 68.
Taylor, I. R. (2003). Australia's temporary wetlands: what determines their suitability as feeding and breeding sites for waders? *Wader Study Group Bulletin* **100**, 54-58.
Thomas, D. (1987). Variations in numbers of Palaearctic waders in south-eastern Tasmania. *The Stilt* **11**, 40-45.
Thomas, D. G. (1970). Wader Migration Across Australia. *The Emu* **70**(4), 145-154.
Thomas, D. G., and Dartnall, A. J. (1971). Ecological aspects of the feeding behaviour of two Calidridine Sandpipers wintering in South-eastern Australia. *Emu* **71**, 20-26.
Thomas, D. G. (1986). Gut contents of four small waders. *The Stilt* **9**, 30-31.
Thomas, D. G. (1988). Pecking rates of waders in south-eastern Tasmania. *The Stilt* **13**, 30-34.
Thomas, K., Kvitek, R., G, and Bretz, C. (2003). Effects of human activity on the foraging behaviour of sanderlings *Calidris alba*. *Biological Conservation* **109**, 67-71.
Thompson, J. J. (1992). Spatial and temporal patterns of shorebird habitat utilisation in Moreton Bay, Queensland. Ph.D. Thesis, University of Queensland, Brisbane.
Thompson, J. J. (1998). Interseasonal Changes in Shorebird Habitat Specialisation in Moreton Bay, Australia. *Emu* **98**, 117-126.
Tockner, K., and Stanford, J. A. (2002). Riverine flood plains: present state and future trends. *Environmental Conservation* **29**(3), 308-330.
Todd, M. K. (2000). Feeding ecology of Latham's Snipe *Gallinago hardwickii*, in the Lower Hunter Valley, New South Wales. *Emu* **100**(2), 133-138.

Tracey, J., Woods, R., Roshier, D., West, P., and Saunders, G. (2004). The role of wild birds in the transmission of avian influenza for Australia: an ecological perspective. *Emu* **104**(2), 109-124.

Truett, J., Miller, M., and Kertell, K. (1997). Effects of Arctic Alaska Oil Development on Brant and Snow Geese. *Arctic* **50**(2), 138-146.

Tulp, I., and de Goeij, P. (1991). Hot news from a hot place: feeding ecology studies on knots and other waders in Broome, northwest Australia, spring 1991. *The Stilt* **19**, 24-26.

Turner, L., Tracey, D., Tilden, J., and Dennison, W. C. (2004). Where river meet sea: Exploring Australia's estuaries. (Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management: Queensland).

Twedt, D. J., Nelms, C. O., Rettig, V. E., and Aycock, S. R. (1998). Shorebird use of managed wetlands in the Mississippi Alluvial Valley. *American Midland Naturalist* **140**(1), 140-152.

Underhill, L., Prys-Jones, R. P., Syroechkovski, J. E. E., Groen, N. M., Karpov, V., Lappo, H. G., van Roomen, M. W., Rybkin, A., Schekkerman, H., Spiekman, H., and Summers, R. W. (1993). Breeding of waders (Charadrii) and Brent Geese *Branta bernicla bernicla* at Pronchishcheva Lake, northeastern Taimyr, Russia, in a peak and a decreasing lemming year. *Ibis* **135**, 277-292.

Underhill, L. G., and Prys-Jones, R. P. (1994). Index numbers of waterbird populations: I. Review and methodology. *Journal of Applied Ecology* **31**, 463-480.

Underwood, A. J. (1992). Beyond BACI: the detection of environmental impacts on populations in the real, but variable, world. *Journal of Experimental Marine Biology and Ecology* **161**, 145-178.

Underwood, A. J. (1997). 'Experiments in Ecology: Their Logical Design and Interpretation Using Analysis of Variance'. (Cambridge University Press: UK).

van de Kam, J., Ens, B. J., Piersma, T., and Zwarts, L. (2004). 'Shorebirds: An Illustrated Behavioural Ecology'. (KNNV Publishers: Utrecht, The Netherlands).

van Gessel, F., W, C, and Kendall, T. (1972). A checklist of the birds of Kooragang Island supplement No. 1. *Hunter Natural History* **4**(4), 256-261.

van Gessel, F. W. C., and Kendall, T. (1972a). A Checklist of the Birds of Kooragang Island. *Hunter Natural History* **4**(3), 194-218.

van Gessel, F. W. C., and Kendall, T. (1972b). The Birds of Kooragang Island: Preliminary Report. *Hunter Natural History* **4**(2), 79-88.

van Gessel, F. W. C. (1973). The Little Greenshank on Kooragang Island. *NSW Birds* **7**(5), 80-81.

van Gessel, F. W. C., and Kendall, T. (1974). A checklist of the birds of Kooragang Island. Supplement No. 2. *Hunter Natural History* **6**(2), 74-86.

van Gessel, F. W. C. (1976). An introduction to some aspects of bird banding and the study of migrant shorebirds on Kooragang Island. *Hunter Natural History* **8**(3), 159-163.

Van Gils, J. A., Piersma, T., Dekinga, A., Spaans, B., and Kraan, C. (2006). Shellfish Dredging Pushes a Flexible Avian Top Predator out of a Marine Protected Area. *PLOS Biology* **4**(12), 2399-2404.

Velasquez, C. R. (1992). Managing Artificial Saltpans as a Waterbird Habitat: Species' Responses to Water Level Manipulation. *Colonial Waterbirds* **15**(1), 43-55.

Velasquez, C. R., and Hockey, P. A. R. (1992). The importance of supratidal foraging habitats for waders at a south temperate estuary. *Ardea* **80**, 243-253.

Warnock, N., Haig, S. M., and Oring, L. W. (1998). Monitoring species richness and abundance of shorebirds in the western Great Basin. *The Condor* **100**, 589-600.

Warren, J. H. (1990). The use of open burrows to estimate abundances of intertidal estuarine crabs. *Australian Journal of Ecology* **15**, 277-280.

Watkins, D. (1993). A National Plan for Shorebird Conservation in Australia. RAOU Report No. 90. (Australasian Wader Studies Group: Melbourne).

Weber, L. M., and Haig, S. M. (1997). Shorebird diet and size selection of nereid polychaetes in South Carolina coastal diked wetlands. *Journal of Field Ornithology* **68**(3), 358-366.

Weishu, H., and Purchase, D. (1987). Migration of Banded Waders between China and Australia. *Colonial Waterbirds* **10**(1), 106-110.

West, A. D., Yates, M. G., McGrorty, S., and Stillman, R. A. (2007). Predicting site quality for shorebird communities: A case study on the Wash embayment, UK. *Ecological Modelling* **202**, 527-539.

West, R. J., Thorgood, C. A., Walford, T. R., and Williams, R. J. (1985). An Estuarine Inventory for New South Wales. Fisheries Bulletin 2. (NSW Fisheries: Cronulla).

Wetlands-International (2006). Waterbird Population Estimates - Fourth Edition. Wetlands International: Wageningen, Netherlands.

Wetlands-International (2008). Existing network sites Available at: http://www.wetlands.org/articlemenu (accessed 18 July 2008).

Whitfield, D. P. (2003a). Redshank *Tringa totanus* flocking behaviour, distance from cover and vulnerability to sparrowhawk *Accipter nisus* predation. *Journal of Avian Biology* **34**, 163-169.

Whitfield, D. P. (2003b). Density-dependent mortality of wintering Dunlins *Calidris alpina* through predation by Eurasian Sparrowhawks *Accipiter nisus*. *Ibis* **145**, 432-438.

Wilkins, S., Keith, D. A., and Adam, P. (2003). Measuring Success: Evaluating the Restoration of a Grassy Eucalypt Woodland on the Cumberland Plain, Sydney, Australia. *Restoration Ecology* **11**(4), 489-503.

Williams, R., Hannan, J., and Balachov, V. (1995). Kooragang Wetland Rehabilitation Project: Fish, Decapod Crustaceans and their habitats. First Interim Report Summer 1993/94. (NSW Fisheries: Cronulla).

Williams, R. J., and Watford, F. A. (1997). Identification of structures restricting tidal flow in New South Wales, Australia. *Wetlands Ecology and Management* **5**, 87-97.

Williams, R. J., Watford, F. A., and Balashov, V. (2000). Kooragang Wetland Rehabilitation Project: History of Changes to Estuarine Wetlands of the Lower Hunter River. NSW Fisheries Final Report Series. Report No. 22. (NSW Fisheries: Cronulla).

Wilson, J. R. (2001). Wader Surveys in the Coorong, South Australia in January and February 2001. *The Stilt* **40**, 38-54.

Wilson, J. R., Nebel, S., and Minton, C. D. T. (2007). Migration ecology and morphometrics of two Bar-tailed Godwit populations in Australia. *Emu* **107**, 262-274.

Wilton, K. (2002). Mangrove and saltmarsh habitat dynamics in selected NSW estuaries. Ph.D. Thesis, Australian Catholic University, Sydney.

Winberg, P. C., Lynch, T. P., Murray, A., Jones, A. R., and Davis, A. R. (2006). The importance of spatial scale for the conservation of tidal flat macrobenthos: An example from New South Wales, Australia. *Biological Conservation* **134**(3), 310-320.

Winning, G. (1991). Western Kooragang Island Vegetation Survey. (Shortlands Wetland Centre: Newcastle).

Winning, G. (1996). Vegetation of Kooragang Nature Reserve and Hexham Swamp Nature Reserve and Adjoining Land. Prepared for NSW National Parks and Wildlife Service. (Shortlands Wetland Centre: Newcastle).

Wolters, M., Garbutt, A., and Bakker, J. P. (2005). Salt-marsh restoration: evaluating the success of de-embankments in north-west Europe. *Biological Conservation* **123**, 249-268.

Woodley, K. (2005). Threats to Shorebirds: Managing Mangrove Expansion in the Firth of Thames, New Zealand. In 'Status and Conservation of Shorebirds in the East Asian-Australasian Flyway; Proceedings of the Australasian Shorebird Conference 13-15 December 2003, Canberra, Australia.' (Ed. P. Straw). pp. 92-93. (Australasian Wader Studies Group and Wetlands International - Oceania.).

Yasue, M., Quinn, J., L, and Cresswell, W. (2003). Multiple effects of weather on the starvation and predation risk trade-off in choice of feeding location in Redshanks. *Functional Ecology* **17**(6), 727-736.

Yasue, M. (2005). The effects of human presence, flock size and prey density on shorebird foraging rates. *Journal of Ethology* **23**, 199-204.

Yates, M. G., Goss-Custard, J. D., McGrorty, S., Lakhani, K. H., Durell, S. E. A. L. V. D., Rispin, W. E., Moy, I., and Yates, T. (1999). Sediment characteristics, invertebrate densities and shorebird densities on banks of the Wash. *Journal of Applied Ecology* **30**(4), 599-614.

Zann, L. P. (1995). Our sea, our future: major findings of the state of the marine environment report for Australia. Report for the Department of the Environment, Sport and Territories, Ocean Rescue 2000 Program. (Great Barrier Reef Marine Park Authority: Townsville, Queensland).

Zedler, J. B. (2000). Progress in wetland restoration ecology. *Trends in ecology and evolution* **15**(10), 402-407.

Zedler, J. B., and Callaway, J. C. (2000). Evaluating the progress of engineered tidal wetlands. *Ecological Engineering* **15**, 211-225.

Zedler, J. B. (2004). Compensating for wetland losses in the United States. *Ibis* **146**, 92-100.

Zharikov, Y. (2002). Feeding Ecology of Shorebirds (Charadrii) Spending the Non-breeding Season on an Australian Subtropical Estuarine Flat. Ph.D. Thesis, Department of Zoology and Entomology, School of Life Sciences, University of Queensland, Brisbane.

Zharikov, Y., and Skilleter, G. A. (2002). Sex-specific intertidal habitat use in subtropically wintering Bar-tailed Godwits. *Canadian Journal of Zoology* **80**, 1918-1929.

Zharikov, Y., and Skilleter, G. A. (2003a). Depletion of benthic invertebrates by bar-tailed godwits *Limosa lapponica* in a subtropical estuary. *Marine Ecology Progress Series* **254**, 151-162.

Zharikov, Y., and Skilleter, G. A. (2003b). Nonbreeding Eastern Curlews *Numenius madagascariensis* Do Not Increase the Rate of Intake or Digestive Efficiency before Long Distance Migration because of an Apparent Digestive Constraint. *Physiological and Biochemical Zoology* **76**(5), 704-715.

Zharikov, Y., and Skilleter, G. A. (2004a). Potential interactions between humans and non-breeding shorebirds on a subtropical intertidal flat. *Austral Ecology* **29**, 647-660.

Zharikov, Y., and Skilleter, G. A. (2004b). Why do eastern curlews *Numenius madagascariensis* feed on prey that lowers intake rate before migration? *Journal of Avian Biology* **35**, 533-542.

Zharikov, Y., and Skilleter, G. A. (2004c). A relationship between prey density and territory size in non-breeding Eastern Curlews *Numenius madagascariensis*. *Ibis* **146**, 518-521.

Zöckler, C., and Lysenko, I. (2000). Water birds on the Edge: First circumpolar assessment of climate change impact on Arctic breeding water birds. UNEP World Conservation Monitoring Centre Biodiversity Series No. 11. (WCMC - World Conservation Press: Cambridge, UK).

Zöckler, C., Delany, S., and Hagemeijer, W. (2003). Wader populations are declining - how will we elucidate the reasons? *Wader Study Group Bulletin* **100**, 202-211.

Zwarts, L., Blomert, A., Ens, B. J., Hupkes, R., and van Spanje, T. M. (1990a). Why do waders reach high feeding densities on the intertidal flats of the Banc D'Arguin, Mauritania. *Ardea* **78**, 39-52.

Zwarts, L., Blomert, A., and Hupkes, R. (1990b). Increase of feeding time in waders preparing for spring migration from the Banc d'Arguin, Mauritania. *Ardea* **78**, 237-256.

APPENDICES

APPENDIX A: Publications related to this thesis

A1

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Observer error in shorebird counts (in review)

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Abstract

Many factors bias counts of shorebirds, potentially introducing errors when estimating temporal trends of local populations. We investigated the effect of viewing platform (ground/ boat-based counts) and species' size (small, medium and large) on the accuracy of observer estimates of migratory shorebird flocks in the Hunter estuary, south-eastern Australia. Observer estimates in the field were compared to real flock sizes estimated from digital photographs taken at the same time as each count. Overall, observer accuracy was high, although there was a tendency for overestimation (2 - 13%). Viewing platform did not affect observer accuracy, but species' size did. We also used photographs of flying flocks to investigate the effect of observer experience, flock size and density on observer accuracy. Flock size had more effect on observer accuracy than flock density. Large flocks were generally underestimated. Inexperienced observers underestimated (-14.6 ± 11.3%) large flocks more than experienced persons (+3.3 ± 7.7%).

Key words: accuracy, Australia, flock size, Hunter River estuary, wader

Introduction

Many counts are often needed to detect trends and fluctuations in waterbird populations (Underhill and Prys-Jones 1994; Kingsford *et al.* 1999). The precision of counts can limit the capacity to detect real changes in populations. Factors affecting precision and accuracy include the viewing platform (van der Meers and Camphuysen 1996; Kingsford 1999); the individual colouration, size and habits of birds (Spearpoint *et al.* 1988; Summers *et al.* 1984); and flock composition, density and size (Erwin 1982; Frederick *et al.* 2003). These factors affect species' detectability. Differences among observers contribute further to variability in counts (Link and Sauer 1998). These differences range from disparities in experience, level of fatigue, observer behavioural traits, visual and hearing acuity and the effect of reinforcement between surveys (Erwin 1982; Bayliss and Yeomans 1990).

Several studies have investigated observer error in counts of waterbirds, including colonies of egrets, ibises and herons (Frederick *et al.* 1996; Rodgers *et al.* 2005), and flocks of geese, ducks (Erwin 1982; Bajzak and Piatt 1990) and seabirds (van der Meer and Camphuysen 1996). Although results varied among studies, the general trend was for observers to underestimate flock sizes and for inexperienced observers to underestimate by the greatest margin. In studies of shorebird counts, observers could underestimate flock sizes by 5% (Hale 1974), 8% (Prater 1979) and 10% (Garnett and Carruthers 1982; Hicklin 1987).

Inevitably with counts collected over many years, survey methodology, frequency of counts, tide states, site coverage and observers can vary within sites. Nevertheless, the critical issue is to investigate the size of errors expected under 'normal' counting conditions, providing a measure of counting error against which to investigate whether long-term trends are real. We investigated the effect of viewing platform and species on estimates of migratory shorebird flocks at two high tide roost sites, in the Hunter estuary, south-eastern Australia. We also used photographs of flying shorebirds to investigate the effect of observer experience, flock size and density on observer accuracy.

Methods

Study site

We did this study in November 2005 in the Hunter estuary (32^{o} 51'S / 151^{o} 46'E), New South Wales (NSW), south-eastern Australia. The Hunter estuary is the most important site for shorebirds in the state of NSW (Smith 1991). Thirty-five migratory shorebird species have been recorded within the estuary (Smith 1991). There are two main day roosts: an artificial sand spit with a shallow lagoon and a large rock wall in the North Arm of the Hunter River. During high tides, the sand spit roost is accessible on land, while the rock wall is only accessible by boat. These roost sites have been monitored monthly since 1999 by the Hunter Bird Observers Club.

Field trial

We used experienced observers in five ground-based and four boat-based surveys of shorebird flocks. These counts mimicked regular surveys conducted by the Hunter Bird Observers Club and the NSW National Parks and Wildlife Service. Three observers counted the total number of birds in each flock with either a telescope (20 – 60X magnification) (ground counts) or binoculars (8 x 30 mm) (boat-based counts). Birds were observed from a distance of 150 - 200 m during ground and 10 - 25 m during boat-based counts. Counts were aborted if additional birds arrived or were flushed from the count area. We also took photographs of each flock with a digital camera (Canon Powershot Pro 1, 7x optical zoom, 8 mega pixels) at the time of counting. To maintain independence between estimates, observers did not confer during counts.

During ground counts, only Eastern Curlews *Numenius madagascariensis* were counted, as this was the only species recorded reliably at the sand spit. This roost site was subject to high disturbance by fishers, walkers and birds of prey and, therefore, counts were often not completed before birds were disturbed, limiting the total number of counts that could be completed (N = 5).

We used a four metre inflatable boat to count shorebirds on the rock wall but were limited to calm conditions (< 10 knots wind speed, Beaufort scale 0 - 1). Several migratory shorebird species were present on the rock wall, permitting comparison among different sized species. Three categories of species were counted in separate sub-flocks along the rock wall: Eastern Curlews (large) (N = 17); godwits (medium) (N = 18); and small sandpipers (N = 11). It was difficult and time-consuming for observers to distinguish between Black-tailed Godwits *Limosa limosa* and Bar-tailed Godwits *Limosa lapponica* so these species were grouped under 'godwits'. Red Knots *Calidris canutus* and Curlew Sandpipers *Calidris ferruginea* were recorded in mixed flocks and were grouped as 'small sandpipers'. Observer estimates of flock sizes were compared to counts taken from photographs of the same flocks. We assumed that the photograph count represented the closest estimate to the actual number of birds present. We used the Global Information System (ESRI ArcView GIS 3.2) to mark individual birds on each digitised photograph. This program summed the number of dots on each photograph to give the total number of birds in each flock. This procedure was repeated three times for each photograph to verify flock estimates.

Experience trial

The second trial complemented the field trials by investigating the effect of observer experience, and flock size and density on observer accuracy. Forty-three volunteers rated their experience in one of four classes; experienced and regular observer of shorebirds (N = 10); some experience with counting shorebirds (N = 11): bird watcher with no experience with shorebird counts (N = 13); or non-bird watcher (N = 9). Observers estimated numbers of Eastern Curlews in flight from 12 photographs projected onto a screen for thirty seconds. The photographs were displayed in a random order with no conferring between observers. We displayed three photographs of each flock size category: small (15 – 25 birds), medium (45 – 79 birds) and large (150 – 200 birds). These flock sizes were commonly observed within the Hunter estuary, which supports about 400 non-breeding Eastern Curlews during summer months (Sept - Mar) (Stuart 2006). In each photograph, individual birds were about the same size and positioned in high density configurations (< five bird lengths from nearest neighbour). We displayed three additional photographs of medium-sized flocks, which had been modified to create low density flocks (> five bird lengths to each nearest neighbour). This allowed a comparison of estimates of different sized flocks (small, medium and large) and flock densities (low and high) by different observers.

Statistical analyses

We investigated stochastic and systematic errors in observer estimates from the field trials. Stochastic error measures variability within a count and can be expressed as the ratio of the standard deviation over the mean presented as a percentage. Systematic error is the difference between the mean estimate of birds and the real flock size determined from photographs (as a percentage of the real flock size) (Rappoldt *et al.* 1985). We used one-way Analysis of Variance (ANOVA) tests (SPSS 2005) to investigate the effect of viewing platform and species on mean systematic errors in counts recorded during the field trials. Two-way ANOVAs were used to investigate the effect of observer experience and flock size on mean systematic errors in estimates of flying flocks. A separate ANOVA was used to examine the effect of flock density. Observer was used as a covariate in these analyses. The difference between an observer estimate and the real flock size was log transformed (log (x+1)) to meet the assumptions of parametric analysis. Means and standard errors are presented throughout this paper.

Results

Overall, viewing platform did not significantly affect the accuracy of estimates, with a low mean systematic error recorded during both ground (4.4 ± 2.0%) and boat-based counts (2.3 ± 3.5%) of Eastern Curlews ($F_1 = 3.3$, $P = 0.076$) (Table A1.1). There was, however, some evidence for underestimation of large flocks during boat-based counts (Fig. A1.1). Although the number of ground counts was limited, the variability among observers (stochastic error) was lower during ground counts (2.1 ± 0.9%) than during boat-based counts (7.8 ± 2.8%) (Table A1.1). Species size affected overall observer accuracy ($F_2 = 31.4$, $P < 0.001$) (Table A1.1; Fig. A1.2). The highest systematic error was observed in counts of godwits, which were overestimated by about 13% (± 5.2) and small sandpipers, which were overestimated by 11% (± 14.9) (Table A1.1). Counts of Eastern curlew flocks were closest to real estimates (2.3 ± 3.5%) but their flocks were smaller than godwit or small sandpiper flocks (Fig. A1.2). There was also greater variability among observer estimates (mean stochastic error) for small sandpipers (26.9 ± 5.7%) than godwits (19.1 ± 2.8%) or Eastern curlews (7.8 ± 2.8%) (Table A1.1).

Observer error was low in estimates of flying flocks from photographs. Experienced observers were more accurate than less experienced observers ($F_3 = 2.9$, $P = 0.034$) but experienced observers overestimated (2 - 3%) total flock sizes, while inexperienced observers underestimated (1 - 3%) (Table A1.2). Flock size had the greatest effect on observer accuracy ($F_2 = 563.73$, $P < 0.001$), with low errors observed for small flocks (15 – 25 birds) (-1.0 ± 0.7%) and

higher errors for medium-sized (45 – 70 birds) (4.7 ± 1.8%) and large flocks (150 – 200 birds) (-7.3 ± 4.4%). This did not seem to be an effect of individual observers ($F_1 = 0.01$, $P = 0.901$) but experience may have had some influence on error rates ($F_6 = 2.0$, $P = 0.072$). Non-bird watchers underestimated large flocks (150 – 200 birds) by more than 14%, while experienced observers were closer to real flock sizes (3.3 ± 7.7%) (Table A1.2). Flock density did not significantly affect observer accuracy ($F_1 = 1.6$, $P = 0.211$), although there was a tendency for low density flocks (> 5 birds apart) to be overestimated (8.4 ± 1.9%) more than high density flocks (< 5 birds apart) (1.0 ± 1.4%) (Table A1.2).

Discussion

Effective shorebird conservation is dependent on a detailed understanding of the status of populations and the variability associated with counting shorebird species. Given the reliance of many shorebird monitoring programs on volunteers, counts by multiple observers are often the most practical solution for monitoring a given estuary over a long period. Observer accuracy was high during ground and boat counts of shorebirds in the Hunter estuary in our study (Table A1.1). We found estimates of the Eastern Curlew population in this study site were reasonably accurate with low systematic and stochastic errors for ground and boat-based counts of each sub-flock (Fig. A1.1). Based on this small study, it is reasonable to expect total counts of Eastern Curlews in the estuary to vary by 2 - 8% between years as a result of observer error and up to 19% for godwits and 27% for smaller sandpipers, such as Red Knots and Curlew Sandpipers. Any larger changes, however, may signify real changes in populations.

Our analysis also confirmed that experienced observers had the highest level of accuracy (Table A1.2). There was, however, a tendency for flock sizes to be overestimated, which contrasts with previous studies where shorebird numbers were underestimated (Hale 1974; Prater 1979; Garnett and Carruthers 1982; Rappoldt *et al.* 1985; Hicklin 1987; Mawhinney *et al.* 1993), but experienced observers can sometimes overestimate large flocks of birds (Erwin 1982). The tendency to overestimate may be a result of the counting technique used by individual observers. Accurate counts of birds are possible when flock sizes are small and each individual bird can be counted directly if flocks number no more than a few hundred birds. Observers, however, are often recommended to count larger flocks of shorebirds in blocks of 10, 20 or more birds (Bibby *et al.* 1992; Eriksson *et al.* 1997). This estimation technique may introduce biases into counts, causing observers to overestimate flock sizes, however, this technique is often the most efficient way to count large flocks of birds.

In the past, there has been much emphasis placed on the underestimation of shorebird flock sizes, for instance, many studies recommend the use of maximum counts in comparing numbers of shorebirds among years (Prys-Jones *et al.* 1994; Underhill and Prys-Jones 1994). There is, however, also a need to control for the effects of overestimation, as a tendency to overestimate numbers may affect our ability to detect negative population change. Furthermore, ignoring observer effects may lead to questionable conclusions concerning changes in shorebird populations, potentially resulting in type II errors and inappropriate management decisions. At the least, reasonable estimates of counting error are essential, even if they cannot necessarily be controlled, so that any trends can be more confidently determined.

Errors in counts of sub-flocks may be less important than overall estimates of a population in an estuary (Erwin 1982; Rappoldt *et al.* 1985). Systematic errors in shorebird counts are of more concern than stochastic errors (Rappoldt *et al.* 1985) but have been the least tested in the field. Using photographs to measure systematic errors does have limitations. It is difficult to detect small and cryptic species in photographs and deriving estimates of real flock sizes is time consuming. In reality, birds often move during counts, which aids in the identification of small species and observers also confer during counts to verify individual estimates and this can reduce the levels of observer error. Although viewing platform did not affect estimates of Eastern Curlew flocks, it may have a greater effect on counts of smaller species in denser flocks. Direct counting is easier with large birds at close range but becomes increasingly difficult with large numbers, small species and great distances (Bibby *et al.* 1992). In this study we were limited to comparing counts of Eastern Curlews, which are large birds that are widely spaced at their roost sites. Systematic errors were lowest during boat-based counts where observers got closer to birds than during ground counts. Godwits (13%) and small sandpipers (11%) were over-estimated to a greater degree during boat-based counts. Both were in dense mixed flocks which increased the levels of systematic and stochastic error (Table A1.1; Fig. A1.2).

Our field trials were conducted in good weather conditions but errors may increase during windy or low light conditions. It is difficult to measure the effect of weather or roost background colour in the field but these factors would also affect the levels of observer error. Further, the effect of different estimation techniques requires more testing in the field. This would increase our knowledge of observer error, permitting better counting perhaps with improved techniques and fewer errors. The relatively low levels of error recorded in this study should encourage regular counters of shorebirds. The challenge in future shorebird monitoring programs is to

verify counts where possible (Rogers *et al.* 2006), while being aware of biases and errors in the counting process. Ultimately measurements of these errors will increase our confidence of trend analyses and provide greater power to detect real changes in shorebird populations.

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References

BAJZAK, D., AND J. F. PIATT. 1990. Computer-aided procedure for counting waterfowl on aerial photographs. Wildlife Society Bulletin 18: 125 - 129.

BAYLISS, P., AND K. M. YEOMANS. 1990. Use of Low-level Aerial Photography to Correct Bias in Aerial Survey Estimates of Magpie Goose and Whistling Duck Density in the Northern Territory. Australian Wildlife Research 17: 1 - 10.

BIBBY, C.J., BURGESS, N.D., AND D.A. HILL. 1992. Bird census techniques. Academic Press: London, UK.

ERIKSSON, A., WESTERENG, L. AND J. QUAYLE. (eds). 1997. British Columbia's standardized inventory methodology for: Shorebirds: Plovers, Oystercatchers, Stilts, Avocets, Sandpipers, Phalaropes and Allies. Ministry of Environment, Lands and Parks, Resource Inventory Branch, Terrestrial Ecosystem Task Force: Victoria, British Columbia.

ERWIN, R. M. 1982. Observer variability in estimating numbers: an experiment. Journal of Field Ornithology 53: 159 - 167.

FREDERICK, P. C., T. TOWLES, R. J. SAWICKI, AND G. T. BANCROFT. 1996. Comparison of aerial and ground techniques for discovery and census of wading bird (Ciconiformes) nesting colonies. Condor 98: 837 - 841.

FREDERICK, P. C., B. HYLTON, J. A. HEATH, AND M. RUANE. 2003. Accuracy and variation in estimates of large numbers of birds by individual observers using an aerial survey simulator. Journal of Field Ornithology 74: 281 - 287.

GARNETT, S., AND I. CARRUTHERS. 1982. Error in aerial surveys of wading birds. The Stilt 3: 5 - 7.

HALE, W. T. 1974. Aerial counting of waders. Ibis 116: 412.

HICKLIN, P.W. 1987. The migration of shorebirds in the Bay of Fundy. Wilson Bulletin 99 (4): 540 - 570.

KINGSFORD, R. T. 1999. Aerial survey of waterbirds on wetlands as a measure of river and floodplain health. Freshwater Biology 41: 425 - 438.

KINGSFORD, R. T., WONG, P. S., BRAITHWAITE, L. W., AND M. T. MAHER. 1999. Waterbird abundance in eastern Australia, 1983-92. Wildlife Research 26: 351 -366.

LINK, W. A., AND J. R. SAUER. 1998. Estimating populations change from count data: application to the North American breeding bird survey. Ecological Applications 8: 258 - 268.

MAWHINNEY, K., HICKLIN, P.W. AND J.S. BOATES. 1993. A re-evaluation of the numbers of migrant Semipalmated Sandpipers, *Calidris pusilla*, in the Bay of Fundy during fall migration. Canadian Field Naturalist 107: 19 - 23.

PRATER, A.J. 1979. Trends in accuracy of counting birds. Bird Study 26: 198 - 200.

PRYS-JONES, R. P., L. G. UNDERHILL, AND R. J. WATERS. 1994. Index numbers for waterbird populations. II. Coastal wintering waders in the United Kingdom, 1970/71-1990/91. Journal of Applied Ecology 31: 481 - 492.

RAPPOLDT, C., M. KERSTEN, AND C. SMIT. 1985. Errors in large-scale shorebird counts. Ardea 73: 13 - 24.

RODGERS, J. A., P. S. KUBILIS, AND S. A. NESBITT. 2005. Accuracy of Aerial Surveys of Waterbird Colonies. Waterbirds 28: 230 - 237.

ROGERS, D. I. , ROGERS, K. G., GOSBELL, K., AND C. HASSELL. 2006. Causes of variation in population monitoring surveys: insights from non-breeding counts in north-western Australia, 2004-05. Stilt 50: 176 - 191.

SMITH, P. 1991. The biology and management of waders (Suborder Charadrii) in NSW. NSW NPWS Species Report, Rep. No. 9.

SPEARPOINT, J.A., B. EVERY AND L.G. UNDERHILL. 1988. Waders (Charadrii) and other shorebirds at Cape Recife, Algoa Bay, South Africa: seasonality, trends, conservation, and reliability of surveys. Ostrich 59: 166 - 177.

SPSS. 2005. SPSS for Windows 14.0. SPSS Inc., 1989-2005: Chicago, Illinois.

STUART, A. 2006. Hunter Region of New South Wales 2005 Annual Bird Report. No. 13. Hunter Bird Observers Club: New Lambton, NSW. pgs. 80.

SUMMERS, R.W., C.V. COARSE, R.W. MEEK, P. MOORE AND M. NICHOLL. 1984. The value of single counts of waders on rocky shores. Wader Study Group Bulletin 41: 7 - 9.

UNDERHILL, L. G., AND R. P. PRYS-JONES. 1994. Index numbers of waterbird populations: I. Review and methodology. Journal of Applied Ecology 31: 463 - 480.

VAN DER MEERS, J., AND C. J. CAMPHUYSEN. 1996. Effect of observer differences on abundance estimates of seabirds from ship-based strip transect surveys. Ibis 138: 433 – 437.

Table A1.1 Mean (± S.E.) stochastic and systematic errors (%) in ground and boat-based counts of Eastern Curlews, and boat-based counts of godwits and small sandpipers (*N* = total number of flocks).

Count type	Shorebird	*N*	Stochastic error (%)[a]	Systematic error (%)[b]
Ground	Eastern Curlew	5	2.1 (0.9)	4.4 (2.0)
Boat	Eastern Curlew	17	7.8 (2.8)	2.3 (3.5)
Boat	Godwits	18	19.1 (2.8)	13.1 (5.2)
Boat	Small sandpipers	11	26.9 (5.7)	11.0 (14.9)

[a]variability among observer estimates (standard deviation/ mean).
[b]difference between the mean of observer estimates and the mean real counts (from photographs) (Rappoldt *et al.* 1985).

Table A1.2 Mean (± S.E.) systematic error (% difference compared to actual numbers; positive indicates overestimation and negative underestimation) of counts made by observers of flying Eastern Curlews from still photographs with different flock sizes (small, medium and large) and flock densities (low and high).

Experience		Flock size			Flock density[b]		Total
Level	N[a]	Small (15 - 25)	Medium (45 - 70)	Large (150 - 200)	Low (> 5 NND)	High (< 5 NND)	
Experienced	9	0.4 (0.7)	4.4 (2.4)	3.3 (7.7)	6.3 (9.0)	2.4 (1.7)	3.1 (4.2)
Some experience	11	- 1.7 (1.2)	6.8 (4.8)	- 3.9 (8.0)	11.4 (11.0)	2.2 (4.1)	2.0 (5.5)
Bird watcher only	13	- 0.8 (1.2)	5.0 (2.6)	- 13.3 (8.0)	8.5 (13.0)	1.5 (2.2)	- 1.0 (4.9)
Non-birdwatcher	8	- 2.1 (2.2)	2.0 (3.5)	- 14.6 (11.3)	6.9 (8.0)	- 2.9 (1.9)	- 3.2 (6.6)
Total	41	-1.0 (0.7)	4.7 (1.8)	- 7.3 (4.4)	8.4 (1.9)	1.0 (1.4)	

[a] N = total number of observers in each observer category
[b] NND = nearest neighbour distance

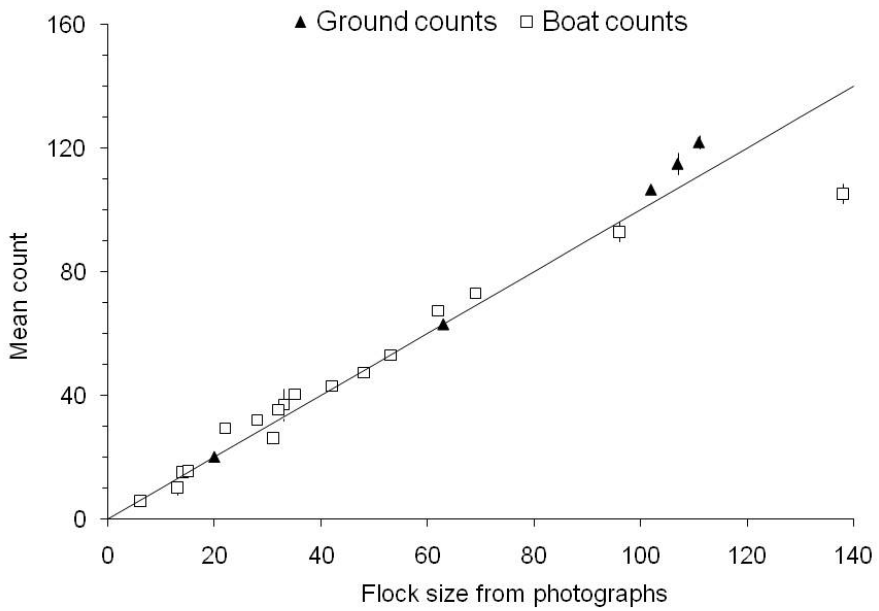

Figure A1.1 Mean (± S.E.) estimates of Eastern Curlew flock size in ground (N = 5) and boat-based counts (N = 17) compared to real flock sizes determined from photographs of the same flocks. Mean counts were close to the expected line of equal flock sizes estimated from photographs.

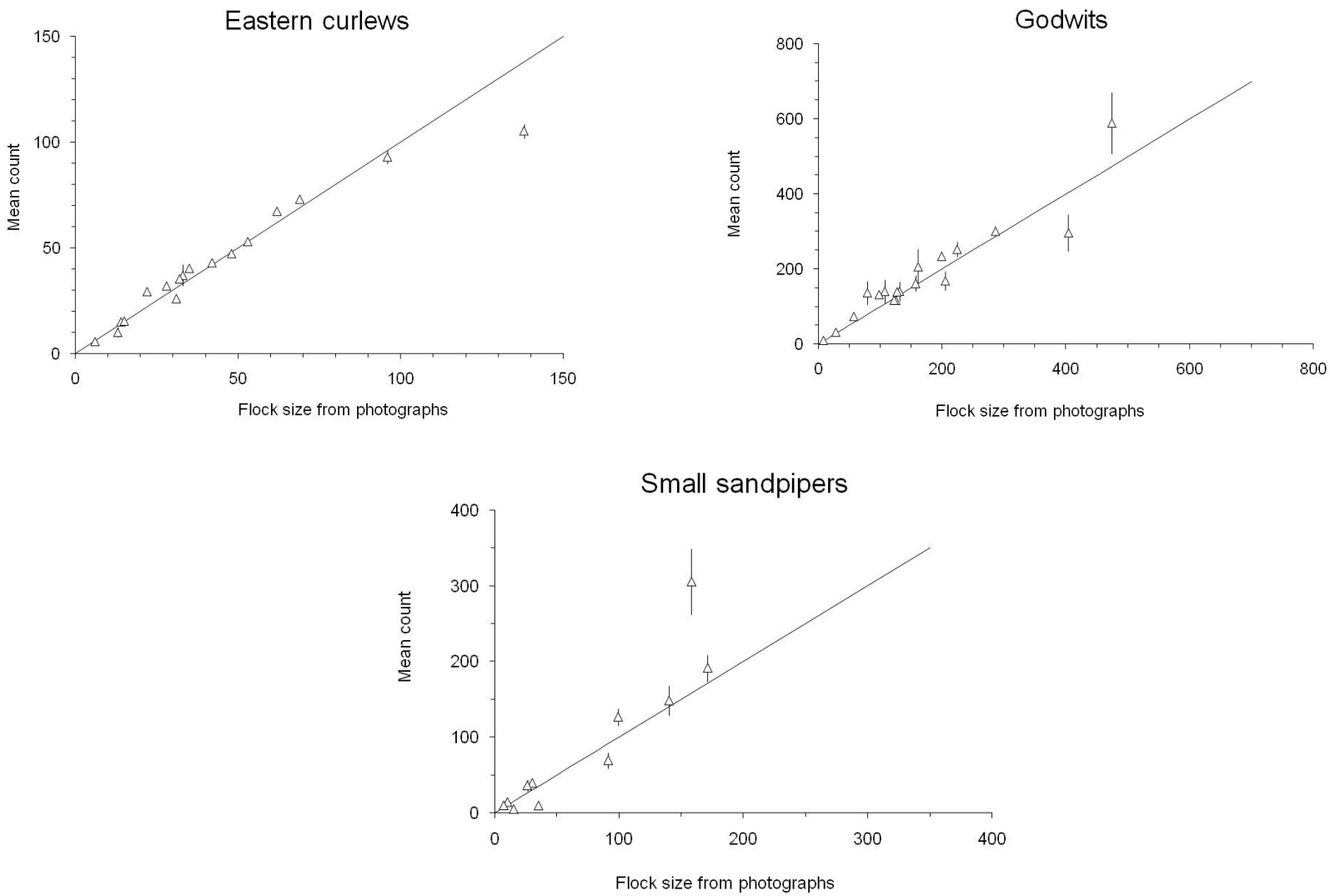

Figure A1.2 Mean (± S.E.) estimates of Eastern Curlew (N = 17), godwit (N = 18) and small sandpiper (N = 11) flocks in boat-based counts compared to real flock sizes determined from photographs of the same flocks. Mean counts of godwits and small sandpipers differed from the expected line, where estimates equal real flock sizes.

A2:

Chapter 7: Saltmarsh as a habitat for birds and other vertebrates

Spencer, J.A., Monamy, V. and M. Breitfuss

Introduction

Saltmarshes are highly productive systems (Adam 1990), yet in comparison with other temperate ecosystems they support relatively few species of terrestrial vertebrates (Greenberg *et al*. 2006). Although the importance of saltmarsh habitat has been documented for faunal species in the northern hemisphere (Sherwood *et al.* 2000; Greenberg *et al*. 2006), and for bird species in particular (Goss-Custard and Yates 1992; Ganter *et al.* 1997; Norris *et al.* 1998; Norris 2000; Hughes 2004), few studies have investigated the importance of saltmarsh habitat for vertebrate species in Australia. This is beginning to change as coastal saltmarsh in Australia gains recognition as important habitat for bird and mammal species (Adam 1990; Morrisey 2000; Laedsgaard 2006). In 2004, for example, coastal saltmarsh was listed as an endangered ecological community in three bioregions in of New South Wales (NSW *Threatened Species Conservation (TSC) Act* 1995) recognising its importance as a feeding and roosting habitat for shorebirds and foraging habitat for insectivorous bats.

Birds

Bird diversity in saltmarsh

Saltmarsh is of direct importance to many avian species by providing habitat in which individuals can breed, feed and roost. In Australia, common colonial waterbird species, such as the Australian White Ibis *Threskiornis molucca*, Straw-necked Ibis *Threskiornis spinicollis* and Cattle Egret *Bubulcus ibis*, can be found in large flocks in coastal wetlands when wetlands in inland Australia are dry (Kingsford and Norman 2002). Large numbers of Black Swans *Cygnus atratus*, Chestnut Teal *Anas castanea* and Australian Shelduck *Tadorna tadornoides* can congregate in saltmarshes in order to feed and roost.

Coastal saltmarsh may also act as drought refuges for Australian breeding shorebird species such as the Black-fronted Dotterel *Elseyornis melanops*, Red-kneed Dotterel *Erythrogonys cinctus,* Black-winged Stilt *Himantopus himantopus* and Red-necked Avocet *Recurvirostra*

novaehollandiae (Lane 1987; Smith 1991) and some species; the Black-winged Stilt; Masked Lapwing *Vanellus miles*; and Red-capped Plover *Charadrius ruficapillus* will also breed in saltmarsh (Marchant and Higgins 1993). Many migratory shorebird species will roost and feed in saltmarsh. These include the Black-tailed Godwit *Limosa limosa,* Common Greenshank *Tringa nebularia*, Curlew Sandpiper *Calidris ferruginea*, Eastern Curlew *Numenius madagascariensis*, Latham's Snipe *Gallinago hardwickii*, Marsh Sandpiper *Tringa stagnatilis*, Pacific Golden Plover *Pluvialis fulva*, Red-necked Stint *Calidris ruficollis* and Sharp-tailed Sandpiper *Calidris acuminata*.

Saltmarsh also provides feeding habitat for terrestrial birds. Birds of prey, such as the Whistling Kite *Haliastur sphenurus* and Swamp Harrier *Circus approximans* are commonly observed over saltmarsh and will hunt small birds and mammals. Small passerines; Zitting Cisticola (Normanton) *Cisticola juncidis normani*, Golden-headed Cisticola *Cisticola exilis*, Little Grassbird *Megalurus gramineus*, Australian Pipit *Anthus novaeseelandiae* and White-fronted Chat *Epthianura albifrons* also frequent saltmarshes. The White-fronted Chat is endemic to Australia and will breed and forage in saltmarsh habitat (Major 1991). Saltmarsh in the wetlands of Sydney Olympic Park, NSW, sustains one of the two remaining populations of White-fronted Chat in Sydney, where their habitat has been lost largely to housing estates and industrial areas (Higgins *et al*. 2001). In south-eastern Australia, this species roosts communally in saltmarsh and nests in samphire shrubs and grass tussocks (Major 1989; Higgins *et al.* 2001).

Australian saltmarshes support many threatened bird species (Table A2.1). For example, saltmarshes in Victoria, Tasmania and South Australia sustain the critically endangered Orange-bellied Parrot *Neophema chrysogaster* (*Environment Protection and Biodiversity Conservation (EPBC) Act 1999*). During its non-breeding season, this parrot feeds on seeds from saltmarsh species: *Frankenia*, *Sarcocornia*, *Sclerostegia* and *Suaeda* (Loyn *et al.* 1986; Orange-bellied Parrot Recovery Team 1998; Garnett and Crowley 2000; Morrisey 2000). The Capricorn subspecies of Yellow Chat *Epthianura crocea macgregori*, which is listed as critically endangered in Australia (*EPBC Act 1999*), nests and forages in saltwater couch grassland and samphire shrubland in central Queensland. This species is only known from Curtis Island, the Torilla Plain and Fitzroy River Delta in central Queensland, but is seasonally mobile and may occur in other locations (Garnett and Crowley 2000). The Slender-billed Thornbill (St Vincents Gulf) *Acanthiza iredalei rosinae* is endemic to samphire shrublands on narrow coastal saline mudflats on northern

shores of Gulf of St Vincent and the Spencer Gulf, in South Australia (Garnett and Crowley 2000).

Although the Bush Stone-curlew *Burhinus grallarius* is usually found in open woodland, in its coastal range it has been observed foraging in saltmarsh (Department of Environment and Conservation NSW 2006). This species has suffered dramatic declines in abundance across southern and eastern Australia and is listed as near threatened under risk criteria developed by the International Union for the Conservation of Nature (IUCN) and as an endangered species in NSW (*TSC Act 1995*) (Table A2.1).

Many threatened waterbird species have been observed feeding in saltmarsh including; Black-necked Stork *Ephippiorhynchus asiaticus*, Black-tailed Godwit, Lewin's Rail *Rallus pectoralis*, Painted Snipe *Rostratula australis* and Radjah Shelduck *Tadorna radjah* (Table A2.1) (Marchant and Higgins 1990; Marchant and Higgins 1993; Higgins and Davies 1996; Garnett and Crowley 2000). Declines in the number and range of many of these threatened species have been associated with the drainage of coastal wetlands (Garnett and Crowley 2000; Olsen and Weston 2004).

Saltmarsh as habitat for migratory shorebirds

Migratory shorebirds, or waders, depend on coastal and inland wetlands and can occur in large numbers (Lane 1987; van de Kam *et al.* 2004). During their migration, shorebird species use many wetland sites, spread across several countries, to sustain their energy supplies before they reach their destination on the breeding or non-breeding grounds. Most migratory shorebird species found in Australia, breed in Alaska, Siberia, Mongolia, northern China and Japan, during June and July of each year, and spend their non-breeding seasons in Australia from September to April (Lane 1987). This migratory route is known as the East Asian-Australasian flyway (Fig. A2.1).

In Australia, little is known of shorebird use of saltmarsh habitats, however, saltmarsh has been documented as important habitat for several shorebird species in South Africa (Puttick 1979; Kalejta 1992; Velasquez and Hockey 1992), North America (Bildstein *et al.* 1982; Erwin *et al.* 1994) and Europe (Goss-Custard and Yates 1992; Norris *et al.* 1998; Rosa *et al.* 2003). Most shorebirds feed on invertebrates in intertidal mudflats and are forced to rest at high tide at roost sites, when their low tide feeding habitat is inundated (Lane 1987). Some species differ from this pattern, by feeding almost continuously in saltmarsh throughout the tidal cycle (Puttick 1979) or

switching between habitats (Long and Ralph 2001), for example, from mudflats to saltmarshes during high tides (Yasue *et al.* 2003).

Species that commonly feed in saltmarsh in Australia, throughout the tidal cycle, include the: Curlew Sandpiper; Marsh Sandpiper; Red-necked Stint; and Sharp-tailed Sandpiper. Smaller flocks of Eastern Curlew and Pacific Golden Plover are often recorded roosting in saltmarsh during daytime high tides (Geering 1995; Loyn *et al.* 2001). These roosts are most important for these species during spring high tides, adverse weather conditions or as a result of disturbance at their main roost sites (Geering 1995; Lawler 1996). At low tides, Eastern Curlew and Pacific Golden Plover usually move to feed in intertidal mudflat rather than remain in saltmarsh continuously.

Shorebird distributions and densities usually match the distribution of their preferred prey species (Goss Custard 1970; 1977; Zharikov and Skilleter 2004). Most species segregate themselves in intertidal habitat according to preferences for sediment penetrability and water depth, as birds prefer to feed in shallow water or wet substrates (Lane 1987). Availability of prey is often determined by the maximum depth at which a shorebird can insert its bill into the substrate and maximum leg length (Dann 1987). This allows a suite of species to co-exist in the same feeding habitat (Lane 1987; Dann 1999). Water depth in saltmarsh habitats is most critical in determining prey availability to shorebird species. The amount of bare substrate or shallow water available is a function of factors such as; water level, topography, water manipulation, local rainfall, soil type and wind action (Skagen & Knopf 1994).

Migratory shorebird species, such as the Sharp-tailed Sandpiper (Fig. A2.2 (a)), feed on invertebrates in the bare substrate fringing low-level saltmarsh vegetation. The Sharp-tailed Sandpiper is the most common migratory shorebird species in NSW (Smith 1991) and has been observed in large flocks of > 1, 000 birds in areas of saltmarsh on Kooragang Island, in the Hunter estuary, NSW. In a study of Sharp-tailed Sandpiper foraging behaviour at this location, 70% of birds were observed foraging during flock scans (n = 124), while a smaller proportion of birds were observed resting and preening (J. Spencer *unpub. data*). The Sharp-tailed Sandpiper is an opportunistic feeder (Higgins and Davies 1996) and was observed feeding extensively on adult chironomids in saltmarsh and high numbers of insect parts were also found in faecal samples examined from this species at this site. Many shorebird species feed extensively on chironomids and this insect biomass has been manipulated in some wetlands to attract shorebirds (Rehfisch 1994).

An additional benefit that may arise for shorebirds feeding in saltmarsh throughout the day is that they can use saltmarsh as night roosting habitat. Small shorebird species, such as the Sharp-tailed Sandpiper, need to feed almost continuously throughout the tidal cycle, and often make extensive use of higher flats (Goss Custard and Moser 1988) or supratidal artificial wetlands (Masero *et al.* 2000; Masero and Perez-Hurtado 2001; Masero 2003) during high tides. By feeding in saltmarsh throughout the day, Sharp-tailed Sandpipers may limit any extra energy that would be expended travelling to and from a separate roost site.

Perceived predation risk is thought to underpin the selection of both feeding and roosting sites by shorebirds (Lawler 1996; Luis *et al.* 2001; Rogers 2003; see Table A2.2). Shorebirds usually select different sites to roost in at night than during the day (Lawler 1996; Sitters *et al* 2001; Rohweder 2001; Rogers 2003) and these can include areas of flooded saltmarsh. Shorebirds are more vulnerable to ground predators, such as foxes, cats and dogs at night (Rogers 2003) but in day-time high tides will select roosts closest to their low tide intertidal feeding habitat and sites which usually have an open aspect allowing easy detection of birds of prey (Lawler 1996; Luis *et al.* 2001; Rogers 2003).

Avian predators are known to be an important cause of mortality in small shorebird species in the Northern Hemisphere (Page and Whitacre 1975; Creswell 1996; Dekker 1998; Hotker 2000; Dekker and Ydenberg 2004). Saltmarsh habitat may provide additional benefits to Sharp-tailed Sandpipers that have non-breeding plumage that is well camouflaged against saltmarsh vegetation. On Kooragang Island, birds were observed flattening themselves against *Sarcocornia quinqueflora* and *Sporobolus virginicus* and moving into this vegetation when birds of prey were overhead or if alarm calls were given.

In the Hunter estuary, NSW, the main night-time roost used by shorebirds is located in flooded saltmarsh on the north-western portion of Kooragang Island. This night roost is characterised by large pools of open water and low-level sparse vegetation dominated by the samphire *Sacrocornia quinqueflora* (Fig. A2.2 (b)). During night-time high tides, this roost can support large flocks of Eastern Curlews (> 100 birds) and Bar-tailed Godwits *Limosa lapponica* (400 – 600 birds) (J. Spencer *unpublished data*). Both species roost on a large sandspit and rock training wall in the lower Hunter River during day-time high tides but these day roosts are deserted at night. Shallow pools of water at the night roost may be attractive to shorebirds as they provide protection from ground predators. Many shorebird species have poor night vision (Rojas

et al. 1999), therefore they may well rely on the noise created by a ground predator moving through open water as a form of predator defence.

This night roost also provides supplementary feeding habitat for shorebirds during night time high tides, including small numbers of Bar-tailed Godwit, Black-tailed Godwit, Common Greenshank, Eastern Curlew and Sharp-tailed Sandpiper. The provision of supplementary feeding habitat is important to migratory shorebirds in their pre-migratory period when they need to gain weight rapidly (Kersten and Piersma 1987; Battley *et al.* 2005) in order to make a successful migration to their northern hemisphere breeding grounds. Some shorebirds also may rely on supplementary feeding at high tide roosts when they have not been successful in meeting their energy requirements in the previous low tide period (Caldow *et al.* 1999; Smart and Gill 2003).

Threats to shorebird populations

Shorebirds face a number of threats to their populations and habitats in the East Asian-Australasian flyway. A total of 20% of shorebird species that use this flyway are listed as critically endangered or near threatened under IUCN risk criteria (Barter 2002). Migratory shorebirds are listed under international migratory bird agreements that Australia has with Japan (JAMBA), China (CAMBA) and the Republic of Korea (ROKAMBA). Under these agreements these countries have obligations to protect migratory shorebirds and their feeding and roosting habitat. At a national level, Australia addresses its obligations through the EPBC Act (Commonwealth of Australia 1999), which contains important provisions for the protection and conservation of shorebirds. Any action that will have, or is likely to have, a significant impact on shorebirds and their habitat requires approval under the Act.

In their non-breeding range, degradation of habitat and excessive disturbance at roost and feeding habitat are thought to be the main threats to migratory shorebird populations (Smith 1991; Watkins 1993; Department of Environment and Heritage 2005). Shorebirds can suffer high disturbance rates at their roosting and feeding sites by recreational users, such as fishers and their watercraft, and walkers and their pets (Burger and Gochfield 1991; Fitzpatrick and Bouchez 1998; Paton *et al.* 2000; Blumstein *et al*. 2003) and by machinery and vehicles used at construction sites (Burton *et al.* 2002; Durell *et al.* 2005). The effect of cumulative disturbance events, from both avian predators and human-induced sources, can result in decreases in energy stores that are needed for moult and migratory fuelling (Burger and Gochfeld 1991). This has implications for

energy conservation as any extra time spent in flight can have significant effects on shorebird body condition and mortality (Durell *et al.* 2005).

In south-eastern Australia, large areas of coastal saltmarsh have historically been drained for agricultural and urban development (Adam 1981; Bucher and Saenger 1991; Zann 1995) and like many habitats saltmarsh is threatened globally by ongoing development pressures and insensitive use (Adam 2002). In some cases, this has directly impacted on the availability of night roosting habitat for shorebirds (Clarke and van Gessel 1983) and caused declines in the numbers of some migratory shorebird species. For example, the loss of saltmarsh may have contributed to declines in the number of Sharp-tailed Sandpipers in NSW (Lane 1987).

Encroachment of mangrove into saltmarsh habitats (Saintilan and Williams 1999) is a major threat to remaining shorebird feeding and roosting habitats (Saintilan 2003; Straw and Saintilan 2005). Most shorebird species prefer open roost sites which allow the detection of potential predators (Lawler 1996; Luis *et al.* 2001; Rogers *et al*. 2006). Therefore, most shorebirds will avoid areas with tall vegetation (Lawler 1996; Rogers *et al.* 2006), such as mangrove, as this vegetation can provide cover for ambushing birds of prey (Dekker and Ydenberg 2004). For example, in North America, success rates for Peregrine Falcons *Falco peregrinus* were highest near the shore zone where falcons used vegetation as a screen before ambushing shorebirds (Dekker 1998; Creswell 1996). Removal of mangrove in shorebird habitat in the Hunter River estuary, NSW, has been successful in restoring habitat for shorebirds but monitoring is required to determine the effectiveness of this technique in the long term.

In coastlines of Asia, Europe, New Zealand and North America saltmarsh and unvegetated mudflats have been invaded by the cord grass *Spartina* spp. (Callaway and Josselyn 1992; Ruiz *et al.* 1997; Neira *et al*. 2006) which often hybridises with local native species (Greenberg *et al.* 2006). In British estuaries, one species of migratory shorebird, the Dunlin *Calidris alpina*, has declined by ~ 50 % in response to the spread of cord grass *Spartina anglica*, which has reduced the availability of intertidal mudflat (Goss-Custard and Moser 1988). Cord grass has also invaded sites in the Tamar estuary, in Tasmania (Adam 1981) and Western Port Bay, Victoria (Western Port Ramsar Information Sheet 1999). Although it was originally introduced for reclamation of land and stabilisation of mudflats (Laedsgaard 2006), invertebrate communities often change following colonisation by hybrid *Spartina* (Hedge and Kriwoken 2000; Neira *et al.*2006) and consequently most shorebirds avoid these areas (Goss-Custard and Moser 1988; Callaway and Josselyn 1992).

Insectivorous Bats

Numerous species of insectivorous bats have been recorded feeding in mangrove forests (Hoye 2002) and it is thought that these species can also use adjacent habitats, such as saltmarsh, as secondary habitat (Laedsgaard *et al.* 2004; Belbase 2005). Bat species tend to feed in open areas of vegetation and can select riparian zones and tracks in dense forests (Law and Chidel 2002; Patriquin and Barclay 2003). They can also range over large distances (Churchill 1998).

The range of the Little North-western Bat *Mormopterus loridae coburgiana* is restricted to mangrove forests and adjacent areas and this species can be found roosting in the upper branches of the mangrove *Avicennia marina* (Churchill 1998; Duncan *et al.* 1999). Bat species which have been recorded in mangrove habitats and are listed as vulnerable in NSW (*TSC* Act 1995) include the Yellow-bellied Sheathtail Bat *Saccolamius flaviventris*, Eastern Freetail Bat *Mormopterus norfolkensis*, Little Bent-wing Bat *Miniopterus australis*, Large Bent-wing Bat *Mi. schreibersii*, Fishing Bat *Myotis adversus* and Greater Broad-nosed Bat *Scoteanax rueppellii* (Hoye 2002).

Few studies document the direct importance of saltmarsh to insectivorous bats (Mills *et al.* 1994; Hoye 2002; Laegdsgaard *et al.* 2004; Belbase 2005). In a study of bats on Kooragang Island, NSW, calls from ten species of insectivorous bats were recorded overflying and/or feeding over saltmarsh (Belbase 2005). These included seven Vespertilionidae species; i.e., Gould's Wattled Bat *Chalinolobus gouldii*, Chocolate Wattled Bat *C. morio*, *Mi. australis*, *Mi. schreibersii*, Little Forest Bat *Vespadelus vulturnus*, Lesser Long-eared Bat/ Gould's Long-eared Bat *Nyctophilus geoffroyi/ N. gouldii* and Greater Broad-nosed Bat *Scoteanax rueppellii* and three species from the Molossidae family; *Mo. norfolkensis*, Little Freetail Bat *Mormopterus* sp. and White-striped Mastiff Bat *Tadarida australis* (Belbase 2005).

In all, 75% of calls recorded over saltmarsh were from *V. vulturnus* and *Mo. norfolkensis* and high levels of feeding activity were recorded for both species (Belbase 2005). Most insectivorous bats are opportunistic feeders (Churchill 1998), therefore these species could be feeding on flying insects from the Orders Homoptera, Hemiptera, Diptera, Coleoptera, Orthoptera, Hymenoptera and Lepidoptera which are commonly found in or over saltmarsh vegetation (Belbase 2005; Laegdsgaard 2006).

Belbase (2005) reported high numbers of feeding buzzes from insectivorous bats which coincided with the emergence of mosquitoes following spring high tides. Coastal saltmarshes provide habitat for a number of mosquito species, the most abundant being *Aedes vigilax* (Diptera:

Culicidae). This mosquito is the most important nuisance-biting pest and vector of arboviruses (e.g. Ross River virus and Barmah Forest virus) in coastal areas of NSW (Russell and Dwyer 2000). The larvae of this mosquito are found in temporary pools in saltmarsh areas following inundation by high tides or rainfall. There is dramatic variability in populations of this mosquito with adults generally living less than three weeks and also dispersing widely (up to 20 km) from larval habitats. Consequently, the abundance of adult *Ae. vigilax* in estuarine wetlands generally varies in four-week cycles where the greatest abundance of adults occurs approximately 10-14 days following initial inundation of the saltmarsh. The magnitude of population change is dependent on a number of environmental and climatic conditions including tide height, rainfall, rain days, temperature, humidity and predator populations.

While mosquitoes may be consumed by insectivorous bats, little is known of the contribution mosquitoes make to the diet of bats that forage over coastal saltmarsh. While it is unlikely that bats represent an effective mosquito management tool, they may play a role in integrated pest management strategies. It is also important to identify the significance of mosquitoes to the diet of bats and, consequently, the impact broad-scale mosquito control (and subsequent reduction of adult mosquito abundance) may have on coastal bat populations.

Water Mouse

The Water Mouse or False Water Rat *Xeromys myoides* is a small native rodent recorded from coastal saltmarsh, mangrove and coastal freshwater wetlands (Fig. A2.3). It is listed as vulnerable under international, Australian and Queensland state legislation (IUCN; Commonwealth of Australia EPBC Act 1999; Queensland *Nature Conservation (Wildlife) Regulation 1994* of the *Nature Conservation Act 1992*). The Water Mouse is distributed in coastal areas of central and south-east Queensland to the Queensland/ New South Wales border (Van Dyck and Gynther 2003; Ball 2004) and the mainland and near-shore islands of the Northern Territory (McDougall 1944; Redhead and McKean 1975; Magnusson *et al.* 1976; Van Dyck 1997; Woinarski *et al.* 2000). This rodent is probably entirely nocturnal, sheltering during the day and between tidal cycles in constructed nesting mounds and natural or artificial hollows. The species will consume grapsid and grapsoid crabs, intertidal crustaceans, pulmonate snails and marine gastropods (Van Dyck 1997) commonly found in intertidal saltmarsh (Breitfuss *et al.* 2004). The most important threats to the Water Mouse are the loss, degradation and fragmentation of freshwater and intertidal wetland communities.

Other vertebrates

Large animals are often only recorded incidentally in saltmarsh. In southern Australia, kangaroos and wallabies are occasional visitors to upper saltmarsh (Adam 1990). The absence of trees and presence of grasses makes saltmarsh habitats attractive to two species of macropods in particular: the Swamp Wallaby *Wallabia bicolor*; and Eastern Grey Kangaroo *Macropus giganteus*. The Eastern Grey Kangaroo has been recorded in NSW saltmarsh on Kooragang Island (P. Svoboda *pers. obs.*) and Cararma Creek, Jervis Bay (N. Saintilan *pers. obs.*). Rabbits, hares, foxes and rats introduced to Australia are occasional visitors to saltmarsh. Effects of grazing by macropods and rabbits in saltmarsh communities are yet to be investigated (Adam 1990).

Reptiles and amphibians are not a normal feature of saltmarsh habitats but they may be more numerous in some brackish situations (Adam 1990). For instance, the vulnerable Green and Golden Bell Frog *Litoria aurea* (EPBC Act 1999) has been recorded in saltmarsh and mangroves areas during periods of high rainfall (P. Svoboda; K. Darcovich *pers. obs.*). Incidental sightings of reptile species in saltmarsh include: Goannas, or Monitor lizards *Varanus* spp; Red-bellied Black Snakes *Pseudechis porphyricus* in upper marsh Baumea; Estuarine Crocodiles *Crocodylus porosus* in higher level flats in the Northern Territory (P. Adam *pers. obs.*); and Eastern Long-necked Turtles Chelodina longicollis in the wetlands of Sydney Olympic Park, in Sydney (K. Darcovich *pers. obs.*). The Cream-striped Shining-skink *Cryptoblepharus virgatus* is also known from saltmarsh samphire forbland near Townsville, Queensland. These reptiles may feed on amphibians, small mammals and other reptiles in saltmarsh.

Conclusions and implications for management

Despite a general deficiency in the number of studies, there is considerable evidence for the importance of coastal saltmarsh as habitat for bird, bat and mammal species in Australia. Coastal saltmarsh is used as foraging habitat by several nationally threatened species including the: Orange-bellied Parrot; Yellow Chat (Capricorn subspecies); Painted Snipe and the Water Mouse. Saltmarsh also provides extensive habitat for numerous shorebird species by supporting: breeding for several resident shorebird species; feeding and roosting habitat for resident and migratory shorebirds; and major night-time roosting habitat for many shorebird species. It also provides secondary feeding habitat for at least ten species of insectivorous bats. It is likely that saltmarsh supplies a range of invertebrate prey items that can be utilised by shorebird and insectivorous bat species. As peripheral wetlands, coastal saltmarshes act as a drought refuge for colonial waterbird

species and Australian breeding shorebirds that are more commonly found in inland Australia. These saltmarshes may become increasing important for these species as a result of increased drought severity and periodicity and river regulation in inland Australia (Kingsford 2000; Olsen and Weston 2004; Kingsford and Porter 2006).

The challenge is how to manage remaining coastal saltmarshes sensitively so that they continue to support a high diversity of plant and animal species. The main threat to species of shorebird and the Water Mouse is the loss, degradation and fragmentation of habitat, either through sea level rise, drainage for urban and industrial developments or physical changes that modify tidal amplitude and frequency of inundation for saltmarsh. The encroachment of mangrove and cord grass into saltmarsh habitats is also a major threat to shorebird feeding and roosting habitats in southern Australia. The removal of standing water by runnelling in order to control mosquito populations may also conflict with the provision of shorebird feeding and roosting habitat. Bats may also be vulnerable to the secondary effects of some insecticides used to control nuisance-biting mosquitoes (Clark 1988). It is also unclear what contribution the saltmarsh mosquito *Aedes vigilax* makes to the diets of shorebird and insectivorous bat species. Ponds are often created within tidal marshes to create waterfowl habitat and to assist in the control of mosquitoes (Erwin *et al.* 1994). Water levels and salinity regimes of these areas need to be managed to maintain a complex mosaic of moist ground and shallow water with sparse vegetation, as these areas support the largest concentrations of shorebird species. By maintaining natural tidal flows and ensuring that wide shallow edges are incorporated into pools and channels in mosquito runnelling (Lawler 1994), a balance between the provision of shorebird habitat and effective mosquito control could be easily achieved.

The next step in the conservation and management of Australian saltmarshes is to address the lack of detailed studies investigating terrestrial vertebrate species use of this habitat and the links between different trophic levels.

References

Adam, P. 1981. Australian Saltmarshes. *Wetlands (Australia)* **1**: 8-10.

Adam, P. 1990. Saltmarsh ecology. Cambridge University Press: United Kingdom.

Ball, D. 2004. Distribution and habitat of the false water rat, *Xeromys myoides* Thomas, 1889 (Rodentia: Muridae) in intertidal areas of central eastern Queensland. *Memoirs of the Queensland Museum* **49**: 487-494.

Barter, M. 2002. Shorebirds of the Yellow Sea: Importance, threats and conservation status. Pages 1-36. Wetlands International Global Series 9, International Wader Studies 12, Canberra, Australia.

Battley, P. F., D. I. Rogers, J. van de Kam, T. Piersma, C. J. Hassell, A. Boyle, and H.-Y. Yang. 2005. How do red knots *Calidris canutus* leave Northwest Australia in May and reach the breeding grounds in June? Predictions of stopover times, fuelling rates and prey quality in the Yellow Sea. *Journal of Avian Biology* **36**: 494-500.

Belbase, S. 2005. Presence and activity of insectivourous bats in saltmarsh habitat at Kooragang Nature Reserve, Newcastle. BSc (Hons) thesis. Australian Catholic University, North Sydney.

Bildstein, K., R. Christy, and P. DeCoursey. 1982. Energy flow through a South Carolina salt-marsh shorebird community. *Wader Study Group Bulletin* **34**: 35-37.

Blumstein, D. T., L. L. Anthony, R. Harcourt, and G. Ross. 2003. Testing a key assumption of wildlife buffer zones: is flight initiation distance a species-specific trait*? Biological Conservation* **110**:97-100.

Breitfuss, M, Connolly, R and P. Dale. 2004, Densities and aperture sizes of burrows constructed by *Helograpsus haswellianus* (Decapoda: Varunidae) in saltmarshes with and without mosquito control runnels. *Wetlands* **24**: 14-22.

Bucher D. and P. Saenger. 1991. An inventory of Australian estuaries and enclosed marine waters: an overview of results. *Australian Geographical Studies* **29**: 370-381.

Burger, J., and M. Gochfeld. 1991. Human activity influence and diurnal and nocturnal foraging of sanderlings (*Calidris alba*). *The Condor* **93**:259-265.

Burton, N. H. K., M. M. Rehfisch, and N. A. Clark. 2002. Impacts of disturbance from construction work on the densities and feeding behavior of waterbirds using the intertidal mudflats of Cardiff Bay, UK. *Environmental Management* **30**:865-871.

Caldow, W. G., J. D. Goss-Custard, R. A. Stillman, S. E. A. L. V. Dit Durell, R. Swinfen, and T. Bregnballe. 1999. Individual variation in the competitive ability of interference-prone foragers: the relative importance of foraging efficiency and susceptibility to interference. *Journal of Animal Ecology* **68**: 869-878.

Callaway, J. C., and M. N. Josselyn. 1992. The introduction and spread of smooth cordgrass (*Spartina alterniflora*) in south San Francisco Bay. *Estuaries* **15**: 218–226.

Churchill, S. 1998. Australian Bats. Reed New Holland Publishers Pty Ltd. Sydney: Australia.

Clark, D.R. Jr. 1988. How Sensitive are Bats to Insecticides? *Wildlife Society Bulletin* **16**: 399-403.

Clarke, C. J., and F. W. C. Van Gessel. 1983. Habitat Evaluation - Birds. pp. 117-144 in J. Moss, editor. *In* An investigation of the natural areas of Kooragang Island, Hunter River. Department of Environment and Planning, Sydney.

Creswell, W. 1996. Surprise as a winter hunting strategy in Sparrowhawks, *Accipiter nisus*, Peregrines, *Falco peregrinus*, and Merlins, *F. columbarius*. *Ibis* **138**: 684-692.

Dann, P. 1987. The feeding behaviour and ecology of shorebirds. in B. Lane, editor. Shorebirds in Australia. Nelson: Melbourne.

Dann, P. 1999. Foraging behaviour and diets of red-necked stints and curlew sandpipers in south-eastern Australia. *Wildlife Research* **27**: 61-68.

Dekker, D. 1998. Over-ocean flocking by Dunlins, *Calidris alpina*, and the effect of raptor predation at Boundary Bay, British Columbia. *Canadian Field-Naturalist* **112**:694-697.

Dekker, D., and R. Ydenberg. 2004. Raptor predation on wintering Dunlins in relation to the tidal cycle. *Condor* **106**:415-419.

Department of Environment and Conservation NSW. 2006. NSW Recovery Plan for the Bush Stone-Curlew *Burhinus grallarius*. DEC, Sydney.

Department of Environment and Heritage. 2005. Background Paper to the Wildlife Conservation Plan for Migratory Shorebirds. The Migratory and Marine Species Section, Australian Government Department of the Environment and Heritage, Canberra.

Duncan, A., Baker, G.B. and N. Montgomery. 1999. The Action Plan for Australian Bats. National Parks and Wildlife, Canberra.

Durell, S., R. A. Stillman, P. Triplet, C. Aulert, D. O. D. Biot, A. Bouchet, S. Duhamel, S. Mayot, and J. D. Goss-Custard. 2005. Modelling the efficacy of proposed mitigation areas for shorebirds: a case study on the Seine estuary, France. *Biological Conservation* **123**:67-77.

Erwin, R.M., J.S. Hatfield, M.A. Howe, and S.S. Klugman. 1994. Waterbird use of saltmarsh ponds created for open marsh water management. *Journal of Wildlife Management* **58**: 516-524.

Fitzpatrick, S., and B. Bouchez. 1998. Effects of recreational disturbance on the foraging behaviour of waders on a rocky beach. *Bird Study* **45**:157-171.

Ganter, B., P. Prokosch, and B. S. Ebbinge. 1997. Effect of saltmarsh loss on the dispersal and fitness parameters of Dark-bellied Brent Geese. Aquatic *Conservation: Marine and Freshwater Ecosystems* **7**:141-151.

Garnett, S. T. and Crowley, G. M. 2000. The Action Plan for Australian Birds 2000. Commonwealth of Australia, Environment Australia, Canberra.

Geering, D. J. 1995. Ecology of migratory shorebirds in the Hunter River estuary. Pages 1-41. Shortlands Wetland Centre Ltd.

Goss-Custard, J. D. 1970. The responses of redshank (*Tringa totanus* (L.)) to spatial variations in the density of their prey. *Journal of Animal Ecology* **39**:91-113.

Goss-Custard, J. D. 1977. The energetics of prey selection by redshank, *Tringa totanus* (L.), in relation to prey density. *Journal of Animal Ecology* **46**:1-19.

Goss-Custard, J. D., and M. E. Moser. 1988. Rates of change in the numbers of Dunlin, *Calidris alpina*, wintering in British estuaries in relation to the spread of *Spartina anglica*. *Journal of Applied Ecology* **25**:95-109.

Goss-Custard, J.D. and M. G. Yates. 1992. Towards predicting the effect of saltmarsh reclamation on feeding birds on the Wash. *Journal of Applied Ecology* **29**: 330-340.

Greenberg, R., J. E. Maldonado, S. Droege, and M. V. McDonald. 2006. Tidal Marshes: A Global Perspective on the Evolution and Conservation of Their Terrestrial Vertebrates. *BioScience* **56**:675-685.

Hedge, P., and L. K. Kriwoken. 2000. Evidence for effects of *Spartina anglica* invasion on benthic macrofauna in Little Swanport estuary, Tasmania. *Austral Ecology* **25**:150-159.

Higgins, P. J. and S. J. J. F. Davies. 1996. Handbook of Australian, New Zealand and Antarctic Birds. Volume 3. Snipes to Pigeons. Oxford University Press: Melbourne.

Higgins, P. J., Peter, J. M. and W.K. Steele. 2001. Handbook of Australian, New Zealand and Antarctic Birds. Volume 5. Tyrant-Flycatchers to Chats. Oxford University Press: Melbourne.

Hotker, H. 2000. When do Dunlins spend high tide in flight? *Waterbirds* **23**:482-485.

Hoye, G. 2002. Pilot Survey of Microchiropteran Bats of the Mangrove Forest of Kooragang Island, the Hunter estuary, New South Wales. Report to Newcastle City Council. Fly By Night Bat Surveys Pty Ltd.

Hughes, R. G. 2004. Climate change and loss of saltmarshes: consequences for birds. *Ibis* **146**:21-28.

Kalejta B. 1992. Time budgets and predatory impact of waders at the Berg River Estuary, South Africa. *Ardea* **80** (3): 327-342.

Kersten, M., and T. Piersma. 1987. High levels of energy expenditure in shorebirds; metabolic adaptations to an energetically expensive way of life. *Ardea* **75**:175-187.

Kingsford, R. T. 2000. Review: Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Austral Ecology* **25**:109-127.

Kingsford, R. T., and F. I. Norman. 2002. Australian waterbirds - products of the continent's ecology. *Emu* **102**:47-69.

Kingsford, R. T., and J. L. Porter. 2006. Waterbirds and wetlands across Eastern Australia. Technical report prepared for the Department of the Environment and Heritage, Canberra.

Laedsgaard P., Monamy V., and N. Saintilan. 2004. Investigating the presence of threatened insectivorous bats on coastal NSW saltmarsh habitat. *Wetlands (Australia)* **22**(1): 29-41

Laegdsgaard, P. 2006. Ecology, disturbance and restoration of coastal saltmarsh in Australia: a review. *Wetlands Ecology and Management* **14**:379-399.

Lane, B. A. 1987. Shorebirds in Australia. Melbourne, Australia, Nelson Publishers.

Law, B. and M. Chidel 2002. Tracks and riparian zones facilitate the use of Australian regrowth forest by insectivorous bats. *Journal of Applied Ecology* **39**: 605-617.

Lawler, W. 1994. Draft Management Manual for Migratory Shorebird Feeding Habitat in New South Wales Estuaries. Pages 1-154. National Parks and Wildlife Service (NSW).

Lawler, W. 1996. Guidelines for Management of Migratory Shorebird Habitat in Southern East Coast Estuaries, Australia. Department of Ecosystem Management. Armidale, NSW, University of New England.

Long, L. L., and J. C. Ralph. 2001. Dynamics of habitat use by shorebirds in estuarine and agricultural habitats in northwestern California. *Wilson Bulletin* **113**:41 - 52.

Loyn, R. H, Lane, B. A., Chandler, C. and G.W. Carr. 1986. Ecology of Orange-Bellied Parrots *Neophema chrysogaster* at Their Main Remnant Wintering Site. *Emu* **86** (4): 195 - 206.

Loyn, R. H., Dann, P. and E. Mulloch. 2001. Important wader sites in the East Asian-Australasian Flyway: 1. Western Port, Victoria, Australia. *The Stilt* **38**: 39-53.

Luis, A., J. D. Goss-Custard, and M. H. Moreira. 2001. A method for assessing the quality of roosts used by waders during high tide. *Wader Study Group Bulletin* **96**:71-73.

Magnusson, W, Webb, G, and J. Taylor. 1976, Two new locality records, a new habitat and a nest description for *Xeromys myoides* Thomas (Rodentia: Muridae), *Journal of Australian Wildlife Research* **3**: 153-157.

Major, R.E. 1989. Population ecology of the White-fronted Chat near Melbourne, Australia. PhD thesis: Monash University, Melbourne.

Major, R.E. 1991. Breeding Biology of the White-fronted Chat *Ephthianura albifrons* in a Saltmarsh near Melbourne. *Emu* **91** (4): 236-249.

Marchants, S., and P. J. Higgins 1990. Handbook of Australian, New Zealand and Antarctic Birds. Volume 1. Ratites to Ducks. Oxford University Press: Melbourne.

Marchant, S., and P. J. Higgins. 1993. Handbook of Australian, New Zealand and Antarctic Birds. Volume 2. Raptors to Lapwings. Oxford University Press: Melbourne.

Masero, J. A., A. Perez-Hurtado, M. Castro, and G. M. Arroyo. 2000. Complementary use of intertidal mudflats and adjacent salinas by foraging waders. *Ardea* **88**:177-191.

Masero, J. A., and A. Perez-Hurtado. 2001. Importance of the supratidal habitats for maintaining overwintering shorebird populations: how redshanks use tidal mudflats and adjacent saltworks in southern Europe. *The Condor* **103**:21-30.

Masero, J. A. 2003. Assessing alternative anthropogenic habitats for conserving waterbirds: salinas as buffer areas against the impact of natural habitat loss for shorebirds. *Biodiversity and Conservation* **12**:1157-1173.

McDougall, W. 1944, An investigation of the rat problem in Queensland canefields, 2 Species and general habits. *Queensland Journal of Agricultural Science* **1**: 48-72.

Mills, D. J., Norton, T. W. and H. E. Parnaby. 1994. The biogeography and conservation evaluation of forest-dwelling, insectivorous bats in south-east Australia. Unpublished report to NSW National Parks & Wildlife Service, Hurstville, NSW.

Morrisey, D. 2000. Saltmarshes. In 'Coastal Marine Ecology of Temperate Australia' (Eds. Underwood, A.J. and Chapman, M. G.). UNSW press: Sydney, Australia.

Neira, C., E. D. Grosholz, L. A. Levin, and R. Blake. 2006. Mechanisms generating modification of benthos following tidal flat invasion by a Spartina hybrid. *Ecological Applications* **16**:1391-1404.

Norris, K., E. Brindley, T. Cook, S. Babbs, C. Forster Brown, and R. Yaxley. 1998. Is the density of redshank *Tringa totanus* nesting on saltmarshes in Great Britain declining due to changes in grazing management? *Journal of Applied Ecology* **35**:621-634.

Norris, K. 2000. The conservation and management of saltmarshes for birds. Pages 341-357 in B. R. Sherwood, B. G. Gardiner, and T. Harris, editors. British Saltmarshes. Forrest Text: UK.

Orange-bellied Parrot Recovery Team. 1998. Orange-bellied Parrot Recovery Plan 1998-2002. Department of Primary Industries, Water and Environment, Hobart.

Olsen, P. and M. Weston. 2004. The State of Australia's Birds 2004. *Wingspan* **14** (4) (Suppl.)

Page, G., and D. F. Whitacre. 1975. Raptor predation on wintering shorebirds. *The Condor* **77**:73-83.

Paton, D. C., M. Ziembicki, P. Owen, B. Hill, and C. Bailey. 2000. Distribution and abundance of migratory waders and their food in the Estuarine areas of the Murray mouth and patterns in the composition of sediments. *The Stilt* **37**:47.

Patriquin, K.J. and M. R. Barclay. 2003. Foraging bats in cleared, thinned and unharvested boreal forest. *Journal of Applied Ecology* **40**: 646-657.

Puttick, G. M. 1979. Foraging behaviour and activity budgets of Curlew Sandpipers. *Ardea* **67**:111-122.

Redhead, T and McKean, J 1975, A new record of the false water rat, *Xeromys myoides* Thomas, 1889 from the Northern Territory of Australia. *Australian Mammalogy* **1**: 347-354.

Rehfisch, M. M. 1994. Man-made lagoons and how their attractiveness to waders might be increased by manipulating the biomass of an insect benthos. *Journal of Applied Ecology* **31**:383-401.

Rogers, D.I. 2003 High-tide roost choice by coastal waders. *Wader Study Group Bulletin* **100**: 73-79.

Rogers, D. I., P. F. Battley, T. Piersma, J. Van Gils, and K. G. Rogers. 2006. High-tide habitat choice: insights from modelling roost selection by shorebirds around a tropical bay. *Animal Behaviour* **72**:563-575.

Rohweder, D.A. 2001. Nocturnal Roost Use by Migratory Waders in the Richmond River Estuary, Northern New South Wales, Australia. *The Stilt* **40**: 23-28.

Rojas, L. M., R. McNeil, T. Cabana, and P. Lachapelle. 1999. Diurnal and nocturnal visual capabilities in shorebirds as a function of their feeding strategies. *Brain Behavior and Evolution* **53**:29-43.

Rosa, S., J. M. Palmeirim, and F. Moreira. 2003. Factors affecting waterbird abundance and species richness in an increasingly urbanized area of the Tagus estuary in Portugal. *Waterbirds* **26**:226-232.

Ruiz, G. M., J. T. Carlton, E. D. Grosholz, and A. H. Hines. 1997. Global Invasions of Marine and Estuarine Habitats by Non-Indigenous Species: Mechanisms, Extent, and Consequences. *American Zoologist* **37**:621-632.

Russell, R.C. & Dwyer, D. (2000) Arboviruses associated with human disease in Australia. *Microbes and Infection* **2**: 1693-1704.

Saintilan, N., and R. J. Williams. 1999. Mangrove transgression into saltmarsh environments in south-east Australia. *Global Ecology and Biogeography* **8**:117-124.

Saintilan N. 2003. Balancing shorebird habitat requirements with mangrove conservation. Pp 15-19 in Straw P. (Ed.) Status and Management of Migratory Shorebirds in Sydney. Sydney Olympic Park Authority.

Sherwood, B. R. Gardiner, B. G. and T. Harris. 2000. British Saltmarshes. Forrest Text: United Kingdom.

Skagen, S. K., and F. L. Knopf. 1994. Migrating shorebirds and habitat dynamics at a prairie wetland complex. *Wilson Bulletin* **106**:91-105.

Straw, P., and N. Saintilan. 2005. Shorebird Habitat Management in Australia - the threat of mangroves. Pages 87-91 *In* P. Straw, editor. Status and Conservation of Shorebirds in the East Asian-Australasian Flyway; Proceedings of the Australasian Shorebird Conference 13-15 December 2003, Canberra, Australia. Australasian Wader Studies Group and Wetlands International - Oceania.

Sitters, H. P., P. M. Gonzalez, T. Piersma, A. J. Baker, and D. J. Price. 2001. Day and Night Feeding Habitat of Red Knots in Patagonia: Profitability Versus Safety? *Journal of Field Ornithology* **72**:86-95.

Smart, J., and J. Gill. 2003. Non-intertidal habitat use by shorebirds: a reflection of inadequate intertidal resources? *Biological Conservation* **111**:359 -369.

Smith, P. 1991. The biology and management of waders (Suborder Charadrii) in NSW. NSW National Parks and Wildlife Service Species Report No. 9.

Straw, P. and N. Saintilan 2005. Shorebird Habitat Management in Australia - the threat of mangroves. Status and Conservation of Shorebirds in the East Asian-Australasian Flyway; Proceedings of the Australasian Shorebird Conference 13-15 December 2003, Canberra, Australia., Australasian Wader Studies Group and Wetlands International - Oceania.

Turner, L., Tracey, D., Tilden, J. and W.C., Dennison. 2004. Where river meet sea: Exploring Australia's estuaries. Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management: Queensland, Australia.

van de Kam, J., B. J. Ens, T. Piersma, and L. Zwarts 2004. Shorebirds: An illustrated behavioural ecology. KNNV Publishers, Utrecht, The Netherlands.

Van Dyck, S 1997, *Xeromys myoides* Thomas, 1889 (Rodentia: Muridae) in mangrove communities of North Stradbroke Island, south-east Queensland. *Memoirs of the Queensland Museum* **42**: 337-366.

Van Dyck, S and I. Gynther. 2003. Nesting strategies of the Water Mouse *Xeromys myoides* in south-east Queensland. *Memoirs of the Queensland Museum* **49**: 453-479.

Velasquez CR. and P. R. Hockey. 1992. The importance of supratidal foraging habitats for waders at a south temperate estuary. *Ardea* **80** (2), 243-253

Watkins, D. 1993. A National Plan for Shorebird Conservation in Australia. Australasian Wader Studies Group, RAOU Report No. 90.

Western Port Ramsar Site Information Sheet 1999. Wetlands International Ramsar Information website: http://www.deh.gov.au/water/wetlands/database/index.html

Whitfield, D. P. 2003. Density-dependent mortality of wintering Dunlins *Calidris alpina* through predation by Eurasian Sparrowhawks Accipiter nisus. *Ibis* **145**:432-438.

Woinarski, J, Brennan, K, Dee, A, Njudumul, J, Guthayguthay, P., and P. Horner. 2000. Further records of the false water rat *Xeromys myoides* from coastal Northern Territory. *Australian Mammalogy* **21**: 221-223.

Yasue, M., Quinn, J.L., and W. Cresswell. 2003. Multiple effects of weather on the starvation and predation risk trade-off in choice of feeding location in Redshanks. *Functional Ecology* **17**: 727-736.

Zann L.P. 1995. Our Sea, Our Future, Major findings of the State of the Marine Environment Report for Australia. Department of Environment, Sport and Territories, Canberra.

Zharikov, Y., and G. A. Skilleter. 2004. A relationship between prey density and territory size in non-breeding Eastern Curlews *Numenius madagascariensis*. *Ibis* **146**:518-521.

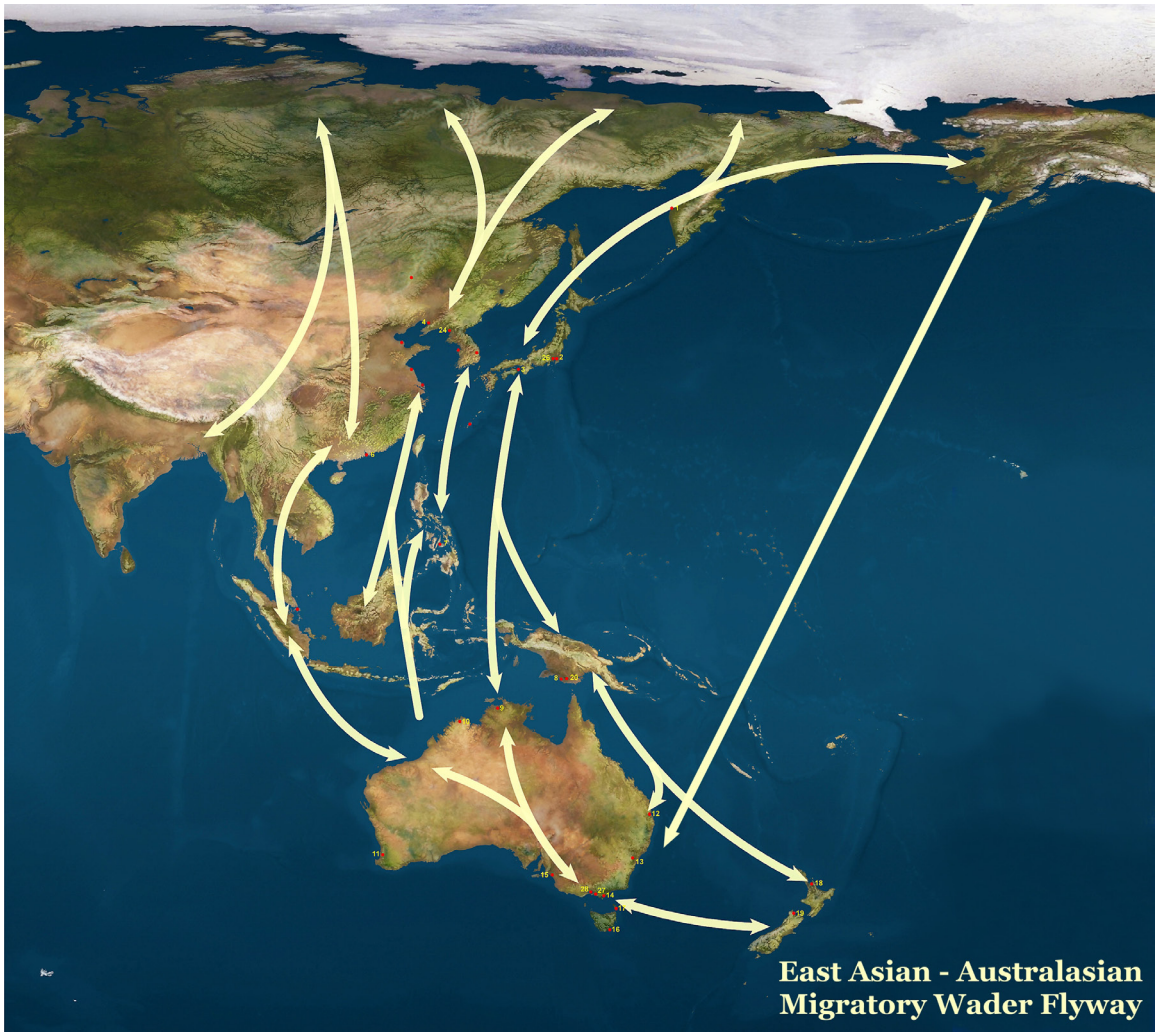

Figure A2.1 The East Asian-Australasian shorebird flyway stretches from non-breeding sites in Australia and New Zealand to breeding sites in Siberia and Alaska (Credit: Australasian Wader Study Group).

(a) **(b)**

Figure A2.2 Flooded saltmarsh on Kooragang Island, the Hunter estuary, New South Wales. Saltmarsh provides (a) important feeding habitat for migratory shorebirds, such as the Sharp-tailed Sandpiper *Calidris acuminata* and (b) major roosting habitat for many species during night-time high tides. (Credit: J. Spencer).

Figure A2.3 The rare Water Mouse or False Water Rat *Xeromys myoides* is a small native rodent recorded from coastal saltmarsh in southern and central Queensland, and the Northern Territory. (Credit: M. Brietfuss).

Table A2.1 Distribution, ecology, status and threats to endangered, threatened and vulnerable bird species reported from Australian saltmarshes.

Species		Distribution	Ecology	IUCN	Status	Main threats
Black-necked Stork	*Ephippiorhynchus asiaticus*	Northern Australia from Pilbara, Western Australia to eastern Queensland to mid New South Wales.	Feed in shallow water, on fish, reptiles & frogs.	NT	e (NSW) r (QLD)	Loss of wetland habitat, disturbance
Black-tailed Godwit	*Limosa limosa*	Migrant, spends non-breeding season in Australia.	Feeds on intertidal mudflats, saltmarshes & brackish wetlands.	NT	v (NSW)	Loss of wetland habitat, disturbance
Bush Stone Curlew	*Burhinus grallarius*	Scarce in southern Australia, remains common in northern Australia.	Usually found on open woodland & grassland but also recorded foraging in saltmarsh. Feeds on invertebrates, reptiles, vegetation & seeds.	NT	e (NSW) e (VIC) v (SA)	Predation by foxes, vegetation clearance
Lewin's Rail (eastern)	*Rallus pectoralis pectoralis*	Mostly coastal, through south eastern Australia; Townsville, through Victoria & Kangaroo Island in South Australia.	Permanent to ephemeral fresh to saline wetlands.	~	v (VIC)	Wetland drainage
Orange-bellied Parrot	*Neophema chrysogaster*	Breeding: south-western Tasmania. Non-breeding: King Island, South Australia and Victoria.	Non-breeding birds disperse to saltmarsh. Feed on seeds of saltmarsh species.	CR	CE	Degradation of grazing habitat
Painted Snipe	*Rostratula benghalensis australis*	Scarce in south-western Australia, range stable in eastern & northern Australia.	Inhabits shallow, vegetated, temporary wetlands, recorded occasionally in saltmarsh. Eats invertebrates & seeds.	~	V	Wetland drainage, vegetation clearance
Radjah Shelduck	*Tadorna radjah*	East Kimberley, Northern Territory to northern Queensland, Cape York Peninsula.	Feeds on small invertebrates & seeds from shallow wetland edges.	~	r (QLD) sp (WA)	Few threats
Rufous Fieldwren (Dirk Hartog Island)	*Calamanthus campestris hartogi*	Dirk Hartog Island, Western Australia.	Low sparse heath, saltmarsh or samphire, feeds on insects, spiders & seeds.	~	r (WA)	Feral animals, fire

Table A2.1 (cont) Distribution, ecology, status and threats to endangered, threatened and vulnerable bird species reported from Australian saltmarshes.

Species		Distribution	Ecology	IUCN	Status	Main threats
Slender-billed Thornbill (St Vincent's Gulf)	*Acanthiza iredalei rosinae*	Gulf of St Vincent, Spencer Gulf, South Australia	Samphire shrublands on narrow coastal saline mudflats. Highly selective of samphire species.	~	v (SA)	Residential, saltworks and marina development
Yellow Chat (Dawson)	*Epthianura crocea macgregori*	Curtis Island, Torilla Plain & Fitzroy River Delta, central Queensland.	Found in freshwater & saline wetlands. Breeds in saltwater couch grassland & samphire shrubland.	~	CE	Wetland drainage, feral pigs, cattle grazing, industrial development
Zitting Cisticola (Normanton)	*Cisticola juncidis normani*	South west Cape York Peninsula, Queensland.	Breeds in saline coastal grasslands.	~	r (QLD)	Few threats

IUCN status: CR = Critically Endangered ; EN = Endangered; NT = Near Threatened.
Australian Status: National (EPBC Act 1999); CE = Critically Endangered; V = Vulnerable.
State listings: e = endangered; v = vulnerable; r = rare; sp = specially protected (NSW = New South Wales Threatened Species Conservation Act 1995; VIC = Victorian Flora and Fauna Guarantee Act 1988; QLD = Queensland Nature Conservation Act 1992; SA = South Australian National Parks and Wildlife Act 1972; WA = Wildlife Conservation Act 1950; NT = Northern Territory Parks and Wildlife Conservation Act 2000).

Table A2.2 High tide roost characteristics and their importance to shorebirds (after Lawler 1996; Luis *et al.* 2001; Rogers 2003).

Roost characteristic	Component type	Importance to shorebirds
Topography/ elevation	Energetics	Protection from adverse weather
Substrate texture and hardness	Energetics	For cooling/ availability of supplementary foraging habitat
Availability of shallow water	Energetics	For cooling/ preening/ availability of supplementary foraging habitat
Proximity to feeding areas	Energetics	Travelling time to and from feeding areas
Vegetation cover/ type	Energetics/ Predation risk	Windbreak/ camouflage from predators
Distance to tall vegetation (visibility)	Predation risk	Tall vegetation provides cover for predators
Proximity to foreshore	Predation risk	Escape distance from predators
Roost background colour	Predation risk	Conspicuousness to predators
Remoteness	Disturbance	Background noise from machinery/ vehicles
Size of roost	Disturbance/ Energetics	Levels of inter/ intra-specific aggression between birds
Distance to alternative roosts	Disturbance/ Energetics	Time spent in flight
Presence of people/ predators	Disturbance/ Predation risk/ Energetics	Time spent in flight

APPENDIX B: List of migratory and vagrant shorebird species recorded in Australia

Common name	Species	Occurrence*
Asian Dowitcher	*Limnodromus semipalmatus*	R
Australian Pratincole	*Stiltia isabella*	R
Baird's Sandpiper	*Calidris bairdii*	V
Bar-tailed Godwit	*Limosa lapponica*	R
Black-tailed Godwit	*Limosa limosa*	R
Broad-billed Sandpiper	*Limicola falcinellus*	R
Buff-breasted Sandpiper	*Tryngites subruficollis*	V
Caspian Plover	*Charadrius asiaticus*	V
Common Greenshank	*Tringa nebularia*	R
Common Redshank	*Tringa tetanus*	R
Common Sandpiper	*Actitis hypoleucos*	R
Curlew Sandpiper	*Calidris ferruginea*	R
Double-banded Plover	*Charadrius bicinctus*	R
Dunlin	*Calidris alpine*	V
Eastern Curlew	*Numenius madagascariensis*	R
Great Knot	*Calidris tenuirostris*	R
Greater Sand Plover	*Charadrius leschenaultii*	R
Grey Phalarope	*Phalaropus fulicaria*	V
Grey Plover	*Pluvialis squatarola*	R
Grey-tailed Tattler	*Heteroscelus brevipes*	R
Hudsonian Godwit	*Limosa haemastica*	V
Latham's Snipe	*Gallinago hardwickii*	R
Lesser Sand Plover	*Charadrius mongolus*	R
Lesser Yellowlegs	*Tringa flavipes*	V
Little Curlew	*Numenius minutus*	R
Little Stint	*Calidris minuta*	V
Little-ringed Plover	*Charadrius dubius*	V
Long-toed Stint	*Calidris subminuta*	R
Marsh Sandpiper	*Tringa stagnatilis*	R
Oriental Plover	*Charadrius veredus*	R
Oriental Pratincole	*Glareola maldivarum*	R
Pacific Golden Plover	*Pluvialis fulva*	R
Pectoral Sandpiper	*Calidris melanotos*	R
Pin-tailed Snipe	*Gallinago stenura*	R
Red Knot	*Calidris canutus*	R
Red-necked Phalarope	*Phalaropus lobatus*	V
Red-necked Stint	*Calidris ruficollis*	R
Ringed Plover	*Charadrius hiaticula*	V
Ruddy Turnstone	*Arenaria interpres*	R
Ruff	*Philomachus pugnax*	R
Sanderling	*Calidris alba*	R
Sharp-tailed Sandpiper	*Calidris acuminata*	R
Spotted Redshank	*Tringa erythropus*	V
Stilt Sandpiper	*Micropalama himantopus*	V
Swinhoe's Snipe	*Gallinago megala*	R
Terek Sandpiper	*Xenus cinereus*	R
Upland Sandpiper	*Bartramia longicauda*	V
Wandering Tattler	*Heteroscelus incanus*	R
Whimbrel	*Numenius phaeopus*	R
White-rumped Sandpiper	*Calidris fuscicollis*	V
Wilson's Phalarope	*Steganopus tricolor*	V
Wood Sandpiper	*Tringa glareola*	R

*Occurrence from Watkins (1993): R = regular, V = vagrant (<5 records annually). Taxonomic nomenclature follows Marchant and Higgins (1993); Higgins and Davies (1996).

APPENDIX C: Review of historical shorebird counts for Ash Island (1993-2007)

To determine the relative importance of Ash Island for migratory shorebird species, I compiled maximum counts of all migratory shorebird species recorded on Ash Island from 1993 to 2007 (Table C1). Anecdotal observations of shorebird species were available in annual reports produced by the NSW Field Ornithologist Club (1994-2002) and the Hunter Bird Observers Club (HBOC) (1993-2007). Low tide surveys of shorebirds on Ash Island were conducted from 1994 to 1997, as part of a larger study of waterbird species in the Hunter estuary (Kingsford *et al.* 1998) (see Chapter 7). Members of the HBOC have counted shorebirds at high tide each month since April 1999. Three separate teams (≥ 2 observers) simultaneously survey nine roost sites in the Hunter estuary (Fig. C1) divided among the survey teams: Ash Island; Deep pond, Long pond, Big pond, Kooragang dykes; and Stockton sandspit, Fern Bay foreshore, Fullerton Cove roost and lower river shoreline (Fig. C1). Sandy and Smith Islands are only surveyed occasionally. Counts are from the ground except for birds roosting on Kooragang dykes and Sandy and Smith Islands, which are counted from a boat. Birds Australia also conducted waterbird surveys in the southern section of Ash Island (wetland sites 12, 13, 14, 15 and 16) (Fig. 7.1) from 2001-02 (Hutchinson & Morris 2003). Low and high tide surveys for shorebirds were carried out as part of my study of Ash Island from 2004-07 (see Chapter 7).

Table C1 Shorebird counts available for Ash Island (1993-2007) (n = number of surveys; NA = information not available).

Source	Time period	Type of data
NSW Bird Observer annual reports (Morris & Burton 1996-99; Morris 2000-04)	1994-2002	Anecdotal observations
Hunter Bird Observers Club annual reports (Stuart 1994-2008)	1993-2007	Anecdotal observations
Kingsford *et al.* (1998)	May 1994-May 1997	Replicate (three) low tide counts: monthly (summer); bi-monthly (winter) (n = 80)
Hunter Bird Observers Club database	Apr 1999-Apr 2007	Monthly high tide surveys (no survey in November 2003) (n = 96)
V. DuBowy, Birds Australia (Hutchinson & Morris 2003)	Jul 2000-Jun 2001	Monthly surveys (n = NA)
A. Hutchinson and A. Morris, Birds Australia (Hutchinson & Morris 2003)	May 2001-Apr 2002	High and low tide counts (n = NA) (irregular, 1-10 surveys/ month)
J. Spencer 2004-06	Nov 2004-Mar 2006	Replicate (three) low tide counts/ month (summer months only) (n = 33)
J. Spencer 2004-07	Nov 2004-Mar 2007	Replicate (three) high tide counts/ month (summer months only) (n = 27)

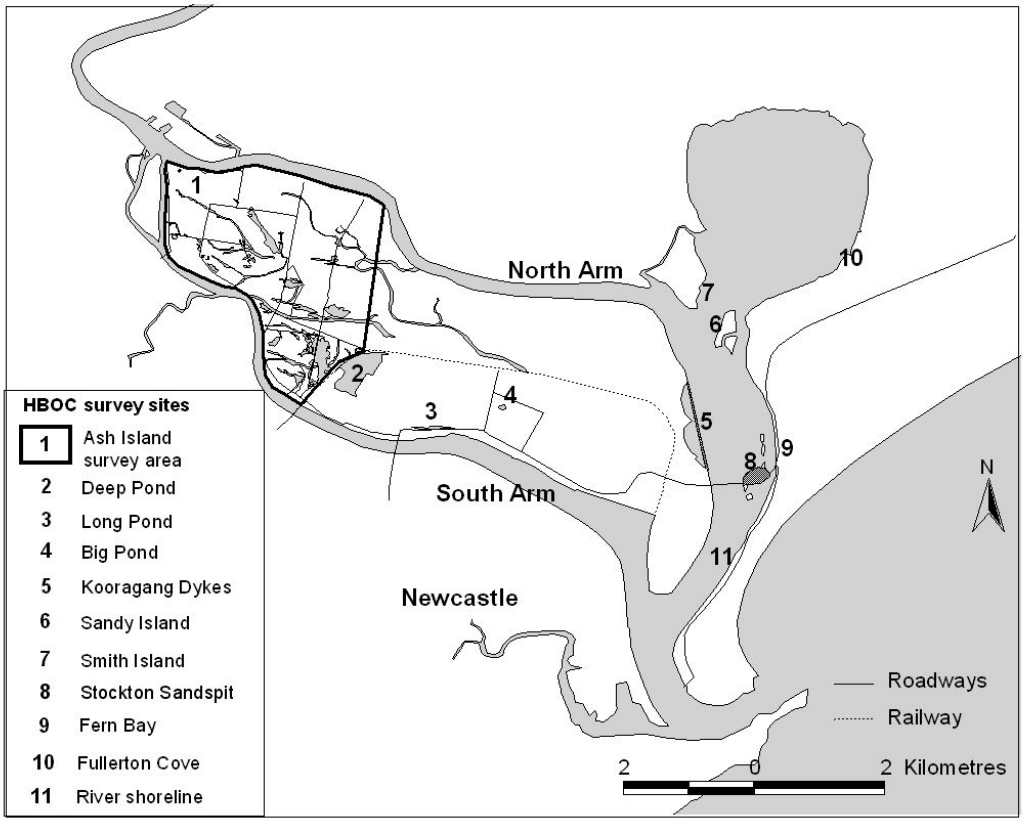

Figure C1 High tide roost sites surveyed each month by members of the Hunter Bird Observers Club (Apr 1999 - Apr 2007).

Ash Island supports a diverse shorebird population, with 21 migratory shorebird species recorded historically (1993-2007) (Table C2). These records include the Black-tailed Godwit *Limosa limosa* and Lesser Sand Plover *Charadrius mongolus*, which are listed as vulnerable under threatened species legislation (*NSW Threatened Species Conservation Act 1995*). Two rare species in Australia, the Ruff *Philomachus pugnax* and Lesser Yellowlegs *Tringa flavipes*, have also been recorded on Ash Island (Table C2). During estuary-wide high tide surveys, total counts on Ash Island represented 10.8% (± 1.4) of total migratory shorebirds found within the Hunter estuary in summer (14 species) (Sep-Apr; n = 64) and 3.2% (± 0.7) in winter (eight species) (May-Aug; n = 32) months (Fig. C2). There were also surveys when counts exceeded 42% of the total counts for the entire Hunter estuary (Apr 2002, Dec 2002, Oct 2006) (Fig. C2), when large flocks of Sharp-tailed Sandpipers *Calidris acuminata* (1,172 birds) and staging Red Knots *Calidris canutus* (1,162 birds) and Red-necked Stints *Calidris ruficollis* (360 birds) were observed foraging on Ash Island. In summer months, four species were observed regularly

during high tide periods and often in large numbers (>100 birds): Common Greenshank *Tringa nebularia* (92% of surveys), Eastern Curlews *Numenius madagascariensis* (86% of surveys), Sharp-tailed Sandpipers (78% of surveys) and Marsh Sandpipers *Tringa stagnatilis* (73% of surveys) (HBOC surveys 1999-2007).

Table C2 Maximum counts of migratory shorebird species in the entire Hunter estuary, compared to Ash Island.

Common name	Hunter estuary[a]	Ash Island	Date[b]	Source[c]
Bar-tailed Godwit	5000	6	Nov-06	HBOC surveys
Black-tailed Godwit	400	6	Sep-06	HBOC surveys
Common Greenshank	362	129	Sep-99	HBOC surveys
Common Sandpiper	6	5	Jan-05	HBOC surveys
Curlew Sandpiper	2637	175	2001-02	Hutchinson and Morris (2003)
Double-banded Plover	180	6	Mar-07	HBOC surveys; Spencer, J. *pers. obs.*
Eastern Curlew	1000	153	Mar-01	HBOC surveys
Grey Plover	4	1	Nov-01	Morris (2003)
Latham's Snipe	20	20	Oct-06	Stuart (2007)
Lesser Sand Plover	170	1	NA	Hutchinson and Morris (2003)
Lesser Yellowlegs	1	1	Sep-01	Stuart (2002)
Little Curlew	7	3	Oct-01	Morris (2003)
Marsh Sandpiper	433	289	Dec-02	HBOC surveys
Pacific Golden Plover	395	300	Jan-Mar 98	Morris (2001)
Pectoral Sandpiper	10	1	Feb-98, Jan-01	Morris (2001, 2003)
Red Knot	2000	1669	Oct-06	Lindsey, A. *pers. ob*.
Red-necked Stint	400	360	Mar-05	Spencer, J. & Lindsey, T. *pers. obs*.
Ruddy Turnstone	50	1	Dec-01	Morris (2003)
Ruff	4	1	Jan-Feb-98, Sep-01, Mar-02, Mar-07	Morris (2001, 2003, 2004); Spencer, J. & Lindsey, T. *pers. obs.*
Sharp-tailed Sandpiper	1711	1711	Mar-05	Spencer, J. and Lindsey, T. *pers. obs*.
Whimbrel	500	4	Mar-05	Spencer, J. *pers. obs*.

[a] Counts based on available counts for the Hunter estuary (1991-2007) (see Chapter 2).
[b] Dates of maximum count for Ash Island (1993-2007) (NA = not available).
[c] Sources of counts for Ash Island.

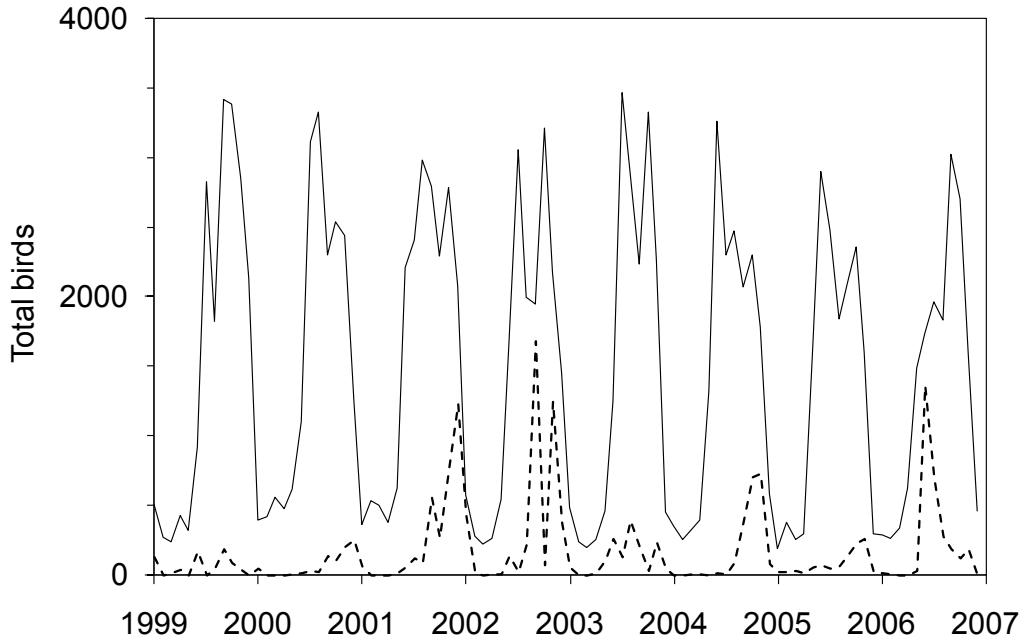

Figure C2 Total numbers of migratory shorebirds recorded on Ash Island (dashed line) relative to all other roosts (solid line) during monthly high tide surveys of the Hunter estuary (HBOC data April 1999 – April 2007; n = 96).

APPENDIX D: Results of benthic invertebrate pilot study (2005)

I conducted a pilot study in December 2005 to determine the most efficient method for sampling benthic invertebrate abundance in intertidal mudflat sites in the Hunter estuary. I collected benthic invertebrates using a PVC corer (90 mm diameter) and estimated crab density by counting their burrows in ten quadrats. Ten cores of three depths (5, 15 and 30 cm) were collected from site 1, in Fullerton Cove (Fig. 5.2) for this pilot study. Counts of the cumulative number of species indicated that five or more cores taken at a depth of 30 cm resulted in the highest number of benthic invertebrate species (Fig. D1). This depth was most effective for collecting large bivalves, however, the coring method tended to underestimate crab abundance. Total numbers of crab burrows were recorded in 1 m^2 quadrats along the water's edge. Mean estimates of crab burrow density were similar for three, five and ten quadrats (Fig. D2).

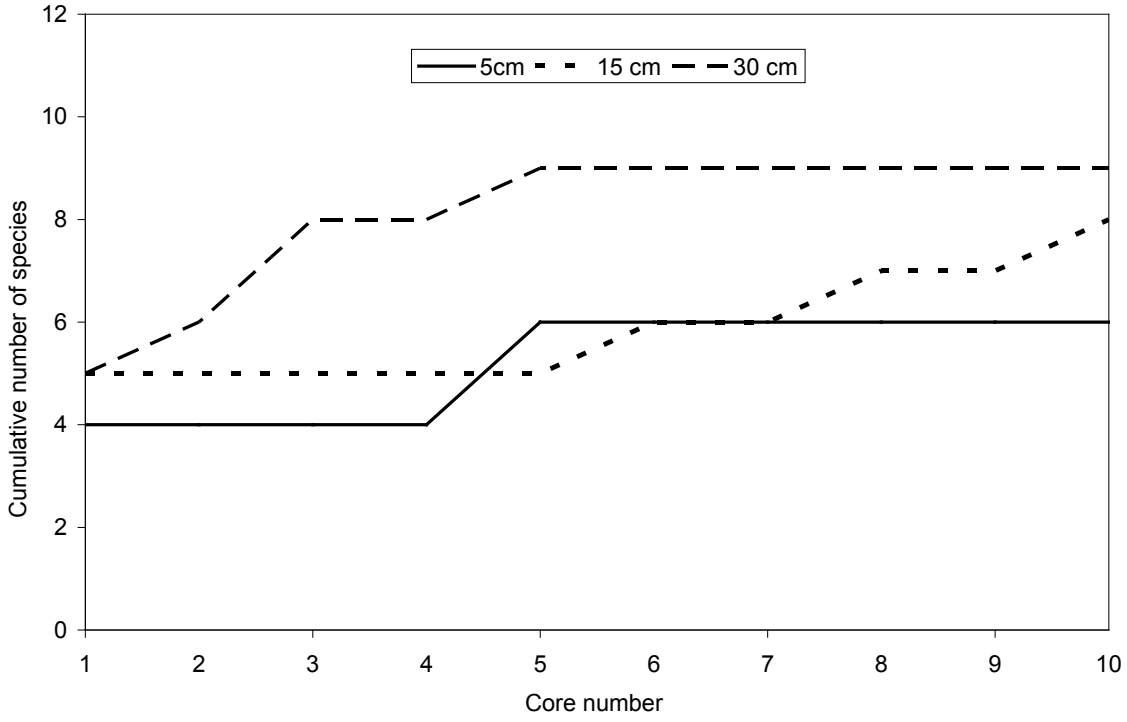

Figure D1 Cumulative number of benthic invertebrate species detected per sampling effort (total number of cores and coring depth). Sampling efficiency was maximized when five to six cores were taken to a depth of 30 cm.

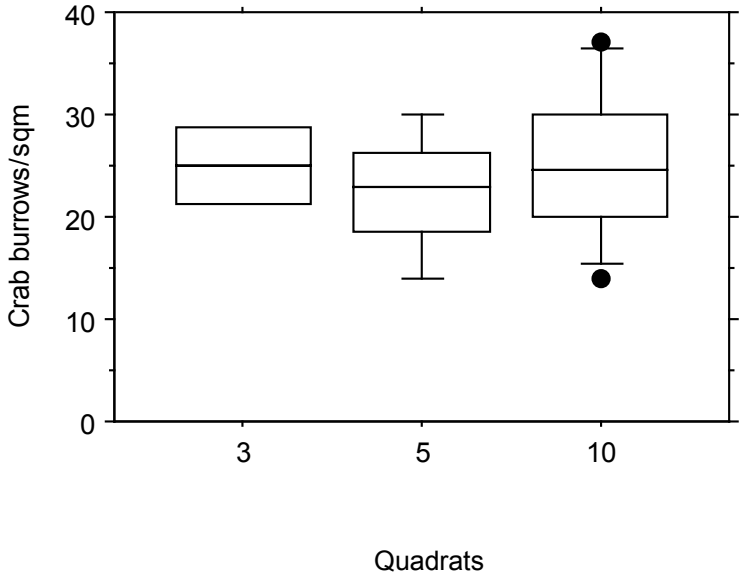

Figure D2 Mean estimates of crab density (burrows/m^2) using three, five and ten quadrats during a pilot survey of an intertidal mudflat in the Hunter estuary.

APPENDIX E: Details of Ash Island surveys (1994-2007)

E1 Dates and tide heights for Ash Island surveys (2004-07) (n = number of surveys).

Low tide surveys (n = 33)		High tide surveys (n = 27)	
Dates	Time/height (m)	Dates	Time/height (m)
20.11.04	9:43/0.68	15.11.04	10:57/1.95
21.11.04	10:57/0.65	16.11.04	11:49/1.90
22.11.04	12:06/0.59	17.11.04	12:45/1.82
18.12.04	8:08/0.64	13.12.04	9:54/2.03
19.12.04	9:18/0.67	14.12.04	10:46/2.03
20.12.04	10:34/0.68	15.12.04	11:39/1.98
03.01.05	8:04/0.72	10.01.05	8:50/2.02
04.01.05	9:11/0.73	11.01.05	9:44/2.07
05.01.05	10:26/0.69	12.01.05	10:35/2.07
15.02.05	8:21/0.64	23.02.05	8:56/1.72
16.02.05	9:31/0.70	24.02.05	9:30/1.75
17.02.05	10:51/0.71	25.02.05	10:03/1.74
03.03.05	8:28/0.60	09.03.05	8:24/1.93
04.03.05	9:49/0.60	10.03.05	9:13/1.93
05.03.05	11:15/0.55	11.03.05	10:00/1.88
10.10.05	6:16/0.65	06.12.06	9:52/1.96
11.10.05	7:33/0.69	07.12.06	10:38/1.91
12.10.05	8:57/0.67	08.12.06	11:22/1.84
24.11.05	8:50/0.80	19.01.07	9:15/1.90
25.11.05	10:02/0.78	20.01.07	10:00/1.95
26.11.05	11:10/0.78	21.01.07	10:44/1.95
21.12.05	6:13/0.69	17.02.07	8:55/1.92
22.12.05	7:04/0.73	18.02.07	9:40/1.96
23.12.05	8:01/0.76	19.02.07	10:25/1.94
22.01.06	8:19/0.73	17.03.07	7:42/1.82
23.01.06	9:30/0.74	18.03.07	8:30/1.88
24.01.06	10:49/0.71	19.03.07	9:16/1.89
19.02.06	6:55/0.62		
20.02.06	7:48/0.66		
21.02.06	8:55/0.69		
22.03.06	8:40/0.63		
23.03.06	10:00/0.61		
24.03.06	11:16/0.54		

E2 Site descriptions and shorebird species recorded in low and high tide surveys of Ash Island (1994-2007) (F/W = freshwater).

Wetland site	Description	Species recorded		
		1994-97[a]	2004-06[b]	2004-07[c]
Wetland 1	Saltmarsh surrounded by mangroves	Common Greenshank Eastern Curlew Pacific Golden Plover Sharp-tailed Sandpiper	None recorded	None recorded
Wetland 2	Pasture and sedges	Common Greenshank Eastern Curlew Marsh Sandpiper	None recorded	None recorded
Wetland 3	Open water surrounded by saltmarsh and Casuarinas	Common Greenshank Curlew Sandpiper Eastern Curlew Marsh Sandpiper Sharp-tailed Sandpiper	None recorded	None recorded
Wetland 4	Saltmarsh surrounded by tall mangroves	Common Greenshank Eastern Curlew	None recorded	None recorded
Wetland 5	Open water, saltmarsh and mangrove	Eastern Curlew Common Greenshank Marsh Sandpiper Sharp-tailed Sandpiper	Common Greenshank	Common Greenshank
Wetland 5 F/W	Pasture with Casuarinas	None recorded	None recorded	None recorded
Wetland 6	Restored saltmarsh	Curlew Sandpiper Common Greenshank	Sharp-tailed Sandpiper	Sharp-tailed Sandpiper
Wetland 6 F/W	Open water in pasture	None recorded	None recorded	None recorded
Wetland 7 F/W	Flooded pasture	Curlew Sandpiper		
Wetland 8	Open water, saltmarsh and mangroves (mangroves removed in 2005)	Common Greenshank Eastern Curlew Marsh Sandpiper Red-necked Stint Sharp-tailed Sandpiper	Common Greenshank Eastern Curlew Sharp-tailed Sandpiper	Common Greenshank Eastern Curlew Sharp-tailed Sandpiper
Wetland 8 F/W	City farm plantation	None recorded	None recorded	None recorded

[a] Low tide surveys 1994-97 (Kingsford et al. 1998); [b] low tide surveys 2004-06; [c] high tide surveys 2004-05 and 2006-07 (J. Spencer).

E2 (cont) Site descriptions and shorebird species recorded in low and high tide surveys of Ash Island (1994-2007) (F/W = freshwater).

Wetland site	Description	Species recorded		
		1994-97[a]	2004-06[b]	2004-07[c]
Wetland 9	Restored saltmarsh, some open water and bare substrate	Common Greenshank Eastern Curlew Marsh Sandpiper Pacific Golden Plover Red-necked Stint Sharp-tailed Sandpiper	Red-necked Stint	Eastern Curlew Red-necked Stint
Wetland 10	Open water surrounded by saltmarsh	Common Greenshank Pacific Golden Plover Marsh Sandpiper Red-necked Stint Sharp-tailed Sandpiper	Common Greenshank Sharp-tailed Sandpiper	Common Greenshank Sharp-tailed Sandpiper
Wetland 11	Open water surrounded by Casuarinas	Common Greenshank Sharp-tailed Sandpiper	Common Greenshank Sharp-tailed Sandpiper	Common Greenshank Marsh Sandpiper Sharp-tailed Sandpiper
Wetland 11 F/W	Flooded pasture	None recorded	None recorded	None recorded
Wetland 12	Flooded saltmarsh, sedges and pasture	Common Greenshank Marsh Sandpiper Sharp-tailed Sandpiper	None recorded	None recorded
Wetland 13	Open water in pasture	None recorded	Common Greenshank	None recorded
Wetland 14	Open water surrounded by saltmarsh and mangroves (mangroves removed in 2005)	Curlew Sandpiper Common Greenshank Eastern Curlew Sharp-tailed Sandpiper Sharp-tailed Sandpiper	Eastern Curlew Sharp-tailed Sandpiper	Common Greenshank Eastern Curlew

[a] Low tide surveys 1994-97 (Kingsford et al. 1998); [b] low tide surveys 2004-06; [c] high tide surveys 2004-05 and 2006-07 (J. Spencer)

E2 (cont) Site descriptions and shorebird species recorded in low and high tide surveys of Ash Island (1994-2007) (F/W = freshwater).

Wetland site	Description	Species recorded		
		1994-97[a]	2004-06[b]	2004-07[c]
Wetland 15	Saltmarsh and pasture surrounded by tall mangroves	Black-tailed Godwit Common Greenshank Curlew Sandpiper Eastern Curlew Marsh Sandpiper Red-necked Stint Sharp-tailed Sandpiper	Eastern Curlew	Common Greenshank
Wetland 16	Large body open water surrounded by saltmarsh (mangroves removed in south of pond in 2005)	Common Greenshank Curlew Sandpiper Eastern Curlew Marsh Sandpiper Pacific Golden Plover Red-necked Stint Sharp-tailed Sandpiper	Common Greenshank Curlew Sandpiper Eastern Curlew Marsh Sandpiper Red-necked Stint Sharp-tailed Sandpiper	Black-tailed Godwit Common Greenshank Curlew Sandpiper Double-banded Plover Eastern Curlew Marsh Sandpiper Pacific Golden Plover Red-necked Stint Ruff Sharp-tailed Sandpiper
Wetland 17 F/W	Flooded pasture/ sedges	Common Greenshank Latham's Snipe Marsh Sandpiper Sharp-tailed Sandpiper	Latham's Snipe	None recorded
Wetland 18	Saltmarsh surrounded by tall mangroves	Common Greenshank Eastern Curlew	Eastern Curlew	Eastern Curlew
Wetland 19	Open water surrounded by saltmarsh	Common Greenshank Pacific Golden Plover Marsh Sandpiper Red-necked Stint Sharp-tailed Sandpiper	Common Greenshank Eastern Curlew Marsh Sandpiper Red-necked Stint Sharp-tailed Sandpiper	Common Greenshank Curlew Sandpiper Eastern Curlew Marsh Sandpiper Pacific Golden Plover Red-necked Stint Sharp-tailed Sandpiper

[a] Low tide surveys 1994-97 (Kingsford et al. 1998); [b] low tide surveys 2004-06; [c] high tide surveys 2004-05 and 2006-07 (J. Spencer).

E2 (cont) Site descriptions and shorebird species recorded in low and high tide surveys of Ash Island (1994-2007) (F/W = freshwater).

Wetland site	Description	Species recorded		
		1994-95[a]	2004-06[b]	2004-07[c]
Wetland 20	River bank and pasture	Not surveyed	Not surveyed	Common Greenshank Common Sandpiper Pacific Golden Plover Whimbrel
Wetland 21	Open water surrounded by Casuarinas	Not surveyed	Not surveyed	Common Greenshank Marsh Sandpiper Sharp-tailed Sandpiper
Wetland 22	Flooded saltmarsh	Not surveyed	Not surveyed	Common Greenshank Sharp-tailed Sandpiper

[a] Low tide surveys 1994-97 (Kingsford et al. 1998); [b] low tide surveys 2004-06; [c] high tide surveys 2004-05 and 2006-07 (J. Spencer).

E3 Mean (± S.E) numbers of migratory shorebird species recorded during low tide surveys in summer months (Oct 1994 - Mar 2006; n = 86).

Survey month	Black-tailed Godwit	Common Greenshank	Common Sandpiper	Eastern Curlew	Latham's Snipe	Marsh Sandpiper	Pacific Golden Plover	Red-necked Stint	Sharp-tailed Sandpiper	Small shorebirds*
Oct-94	0.0 (-)	3.0 (1.5)	0.0 (0.0)	1.3 (0.3)	0.0 (-)	42.0 (13.6)	0.0 (-)	0.7 (0.3)	4.0 (1.5)	3.3 (2.8)
Nov-94	0.0 (-)	17.7 (4.7)	4.0 (2.6)	3.0 (0.6)	0.0 (-)	33.0 (8.0)	0.0 (-)	37.0 (9.1)	10.7 (3.2)	0.3 (0.3)
Dec-94	0.0 (-)	20.0 (6.7)	6.7 (6.2)	3.3 (0.7)	0.0 (-)	127.3 (47.4)	0.0 (-)	5.0 (0.6)	287.7 (53.5)	54.0 (29.1)
Jan-95	0.7 (0.7)	34.0 (3.1)	23.7 (4.7)	3.7 (0.3)	0.0 (-)	149.3 (9.4)	7.0 (7.0)	6.3 (4.1)	126.7 (55.9)	15.7 (11.1)
Feb-95	0.0 (-)	6.3 (0.9)	0.0 (-)	2.0 (1.2)	0.0 (-)	73.3 (36.8)	11.3 (2.4)	1.7 (0.9)	18.0 (18.0)	0.0 (-)
Mar-95	0.0 (-)	3.0 (0.6)	0.0 (-)	14.0 (11.1)	0.0 (-)	40.3 (22.8)	35.3 (18.3)	0.0 (-)	11.3 (11.3)	0.0 (-)
Oct-95	0.0 (-)	12.7 (6.7)	0.0 (-)	1.7 (0.3)	0.0 (-)	0.0 (-)	0.0 (-)	7.7 (3.9)	0.7 (0.7)	0.7 (0.7)
Nov-95	0.0 (-)	11.7 (2.9)	0.0 (-)	16.3 (8.1)	0.0 (-)	11.0 (4.9)	0.0 (-)	30.3 (9.8)	2.7 (1.5)	12.0 (7.6)
Dec-95	0.0 (-)	20.7 (1.7)	0.0 (-)	5.3 (1.9)	0.0 (-)	100.0 (37.7)	0.0 (-)	2.7 (2.7)	29.3 (24.9)	0.3 (0.3)
Jan-96	0.0 (-)	1.0 (0.6)	0.0 (-)	9.3 (1.2)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)
Feb-96	0.0 (-)	1.7 (0.7)	0.0 (-)	12.0 (7.6)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)
Mar-96	0.0 (-)	1.0 (0.6)	0.0 (-)	3.7 (0.3)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)
Oct-96	0.0 (-)	3.7 (3.7)	0.0 (-)	2.3 (0.3)	1.3 (0.9)	0.7 (0.7)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)
Nov-96	0.0 (-)	0.0 (-)	0.0 (-)	0.3 (0.3)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)
Dec-96	0.0 (-)	2.3 (1.9)	0.0 (-)	3.0 (1.0)	0.0 (-)	41.7 (1.7)	0.3 (0.3)	0.3 (0.3)	116.3 (102.0)	0.0 (-)
Jan 97	0.0 (-)	8.3 (5.3)	0.0 (-)	7.7 (2.2)	0.0 (-)	16.3 (7.0)	4.0 (4.0)	0.0 (-)	115.3 (102.4)	0.0 (-)
Feb 97	0.0 (-)	6.7 (4.7)	0.0 (-)	10.7 (2.6)	0.0 (-)	20.3 (20.3)	5.0 (2.5)	0.0 (-)	0.0 (-)	0.0 (-)
Mar 97	0.0 (-)	12.5 (8.5)	0.0 (-)	5.5 (0.5)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)	0.0 (-)
Nov-04	0.0 (-)	2.7 (0.9)	0.0	1.0 (1.0)	0.0	0.0 (0.0)	0.0 (-)	0.0	14.7 (10.9)	0.0
Dec-04	0.0 (-)	10.7 (1.8)	0.0	8.3 (0.9)	0.0	9.0 (4.6)	0.0 (-)	0.0	83.3 (11.4)	0.0
Jan-05	0.0 (-)	17.7 (11.3)	0.0	5.3 (1.8)	0.0	39.0 (9.5)	0.0 (-)	13.7 (3.9)	448.7 (58.5)	0.0
Feb-05	0.0 (-)	8.0 (2.3)	0.0	7.3 (0.7)	0.0	0.0 (0.0)	0.0 (-)	1.7 (1.7)	63.0 (32.9)	0.0
Mar-05	0.0 (-)	15.3 (7.2)	0.7 (0.7)	14.7 (7.5)	0.0	64.7 (17.9)	0.0 (-)	130.3 (115.1)	1120.3 (279.2)	0.0
Oct-05	0.0 (-)	3.0 (2.1)	0.0	3.3 (0.9)	1.0 (1.0)	2.7 (1.3)	0.0 (-)	0.3 (0.3)	122.7 (18.8)	0.3 (0.3)
Nov-05	0.0 (-)	13.0 (0.6)	2.3 (1.9)	2.3 (1.3)	0.0	45.3 (16.7)	0.0 (-)	0.0	7.0 (6.5)	0.0
Dec-05	0.0 (-)	5.7 (2.9)	0.0	5.0 (1.2)	0.0	0.0 (0.0)	0.0 (-)	0.0	0.0 (0.0)	0.0
Jan-06	0.0 (-)	12.3 (6.5)	0.0	1.3 (0.7)	0.0	6.7 (3.4)	0.0 (-)	0.3 (0.3)	33.0 (21.0)	0.0
Feb-06	0.0 (-)	5.0 (1.2)	0.0	4.3 (0.9)	0.0	55.7 (21.3)	0.0 (-)	0.0	233.0 (95.4)	0.0
Mar-06	0.0 (-)	8.3 (1.3)	0.0	1.0 (0.6)	0.0	15.3 (7.9)	0.0 (-)	0.0	7.3 (5.0)	0.0

* unidentified small shorebird species